Buffer Zones:

THEIR PROCESSES AND POTENTIAL IN WATER PROTECTION

Buffer Zones:
Their Processes and Potential in Water Protection

The Proceedings of the International Conference on Buffer Zones

September 1996

Edited by:

Nick Haycock
Haycock Associates Limited, UK

Tim Burt
University of Durham, UK

Keith Goulding
IACR – Rothamsted, UK

Gilles Pinay
CNRS, Université de Rennes, France

Haycock Associated Limited

11 Edison Close, St. Albans, Herts. AL4 0RB

Buffer Zones: Their Processes and Potential in Water Protection

First Published by Quest Environmental, PO Box 45, Harpenden, Hertfordshire, AL5 5LJ, UK (Business changed names to Haycock Associated Limited, April 2000)

Electonic PDF Version Published by Haycock Associated Limited March 2001

British Library Cataloguing in Publication Data
A catalogue record for this book is available from the British Library
ISBN: 0 9530051 0 0

Indexes, proof reading and figures: Quest Environmental (1997)
Design and origination: Mac Graphics, Harpenden, Herts., UK
PDF Version Haycock Associated Limited, Herts., UK
Printing and binding: White Crescent Press Ltd, Luton, Beds., UK

Cover photograph: The River Save, a tributary of the River Garonne, Toulouse, by Nick Haycock

Acknowledgements

These proceedings were financially supported by Quest Environmental and the Environmental Agency R&D Programme on "Land Management Techniques". The Environment Agency continues to be actively involved with the scientific and management issues associated with buffer zones. This book builds on the Environment Agency's booklet "Understanding Buffer Strips", 1996 (available from all Environment Agency regional offices) and previous publications on this subject, notably Large, A.R.G. and Petts, G.E., 1992, *Buffer zones for conservation of rivers and bankside habitats,* Project 340/5/Y. This and other Environment Agency R&D publications are available from the Foundation for Water Research, Allen House, The Listons, Listons Road, Marlow, Bucks, SL7 1FD, UK.

The members of the conference steering group are acknowledged for focusing the debate and securing funds, namely: Kay Schofield and Richard Smith, Environment Agency; Rob Jarman, The National Trust; Richard Knight, FWAG; Mary Gibson, English Nature; Rex Agg, FWR; Paul José, RSPB; Caroline Drummond, LEAF and finally Art Gold, Univ. of Rhode Is., USA and Wendell Gilliam, NCSU, USA.

For those who attended the conference, we are sure you would appreciate us thanking both The Earl of Cranbrook, Chairman of English Nature, for his opening address and Jonathan Peel, Deputy Chairman of the National Trust and Chairman of the Broads Authority, for his conference dinner speech. Thanks also go to: Peter Leeds-Harrison, Cranfield University; Richard Vivash and Martin Janes, River Restoration Project; Andy Mayled and Richard Morris, The National Trust; Rupert Burr, Roves Farm and Ian Bowyer, The Radnor Arms, for their involvement in the conference tours. We would also like to thank the loyal band of conference helpers. Finally a special thanks goes to Val Mitchell and the conference centre staff at the National Westminster Bank Staff Training College, Heythrop.

We would like to thank all the authors of the oral and poster presentations for their work. A selection of the poster presentations is included in this book but a full listing of the conference posters can be found in Appendix I.

Thanks go to the Ministry of Agriculture, Fisheries and Food and ADAS for permission to publish "The Establishment of Buffer Zones – The Habitat Scheme Water Fringe Option" by A. Tytherleigh. We also acknowledge the US Environmental Protection Agency, Chesapeake Bay Program, for permission to use figures 2, 6 and 7 from EPA 903-R-95-004 (see chapters by Dillaha and Inamdar, and also Lowrance). Permission from ASA-CSSA-SSSA was given for figures 2 and 4 in the chapter by Gold and Kellogg.

In the creation of this book, thanks must go to all our authors and also to Laura Haycock, for compiling their many excellent contributions into published form. I am sure we are all grateful for the efficiency of the editors which has enabled us to publish this book so soon after the conference.

Contents

List of Contributors

T. Addiscott, IACR-Rothamsted, Harpenden, Herts., AL5 2JQ, UK

B. Alker, Soil, Waste and Groundwater Group, WRc plc, Medmenham, Henley Road, Marlow, UK and Department of Civil Engineering, Imperial College London, College Road, London, UK

A. Bartoszewicz, Research Centre for Agricultural and Forest Environments, Polish Academy of Science, ul. Bukowska 19, 60809 Poznan, Poland

J. Baudry, National Institute for Agronomic Research, Sad-Armorique Research Unit, 65 rue de Saint-Brieuc, 35042 Rennes Cedex, France

S.D. Bridgham, Department of Biological Sciences, University of Notre Dame, Notre Dame, IN 46556, USA

T.P. Burt, Department of Geography, University of Durham, Science Laboratories, South Road, Durham, DH1 3LE, UK

D. Butler, Department of Civil Engineering, Imperial College London, College Road, London, UK

R.J. Cook, Morley Research Centre, Morley, Wymondham, Norfolk, NR18 9DB, UK

A.B. Cooper, National Institute of Water and Atmospheric Research, Hamilton, New Zealand

D.L. Correll, Smithsonian Environmental Research Center, PO Box 28, Edgewater, MD 21037, USA

F. Díaz-Fierros, Department Edafoloxía e Química Agrícola, Fac. Farmacia, Campus Univ. S/N, 15706 Santiago de Compostela, Spain

T.A. Dillaha III, Department of Biological Systems Engineering, Virginia Polytechnic Institute and State University, Blacksburg, VA 24061 – 0303, USA

B.C. Dickson, Confluence Ecological, PO Box 292, Reno, PA 16343, USA

M.T. Downes, NIWA, PO Box 8602, Christchurch, New Zealand

P. Durand, Institut National de la Recherche Agronomique (INRA), Unité de Science du Sol et de Bioclimatologie, 65 route de saint Brieuc, 35042 Rennes Cedex, France

M. Fagot, Ministry of the Environment, Water Directorate, Paris, France

S. Fleischer, County Administrative Board, S 301 86 Halmstad, Sweden

A. Forster, Huntingdon Life Sciences, PO Box 2, Huntingdon, Cambridgeshire, PE18 6ES, UK

J.L. Gardiner, Department of Geography and Environmental Management, Middlesex University, Queensway, Enfield, London, EN3 4SF, UK

J.W. Gilliam, North Carolina State University, Department of Soil Science, Box 7619, Raleigh 27695-7619, USA

J. Girel, UPR-ES 592 Laboratoire Ecosystèmes Alpins, Groupe Hydrosystèmes Alpins, Université Joseph Fourier, Centre de Biologie Alpine, B.P. 53, 38041 Grenoble (cedex 9), France and ESA C.N.R.S. 5023 (Ecologie des Eaux douces et des Grands Fleuves)

A.J. Gold, Department of Natural Resources Science, University of Rhode Island, Kingston, RI 02881, USA

K.W.T. Goulding, IACR-Rothamsted Experimental Station, Harpenden, Herts., AL5 2JQ, UK

J.J. Gril, Institut de recherche pour l'ingéniérie de l'agriculture et de l'environnement (Cemagref), Water Quality Division, 3 bis quai Chauveau CP110, F-69336, cedex 09, Lyon, France

P.M. Groffman, Institute of Ecosystem Studies, Box AB, Millbrook, NY 12545, USA

B. Gumiero, Department of Biology, University of Bologna, Via Selmi. 3, Bologna 40126, Italy

G.L. Harris, ADAS Land Research Centre, Meden Vale, Mansfield, Notts., NG20 9PF, UK

H. Hartikainen, University of Helsinki, Department of Applied Chemistry and Microbiology, Finland

N.E. Haycock, Quest Environmental, PO Box 45, Harpenden, Herts., AL5 5JL, UK

A.R. Hill, Department of Geography, York University, 4700 Keele Street, North York, Ontario, M3J 1P3, Canada

C. Howard-Williams, NIWA, PO Box 8602, Christchurch, New Zealand

S.P. Inamdar, USDA-ARS Southeast Watershed Research Laboratory, PO Box 946, Tifton, GA 31793, USA

M. Ivask, Environmental Protection Institute, Estonian Agricultural University, 5 Kreutzwaldi St. EE-2400 Tartu, Estonia

A. Joelsson, County Administrative Board, S 301 86 Halmstad, Sweden

C. A. Johnston, Natural Resources Research Institute, University of Minnesota, 5031 Miller Trunk Highway, Duluth, MN 55811, USA

A. Kedziora, Research Centre for Agricultural and Forest Environments, Polish Academy of Science, ul. Bukowska 19, 60809 Poznan, Poland

D.Q. Kellogg, Department of Natural Resources Science, University of Rhode Island, Kingston, RI 02881, USA

V. Kuusemets, Institute of Geography, University of Tartu, 46 Vanemuise St., EE2400 Tartu, Estonia

K. Lõhmus, Institute of Geography, University of Tartu, 46 Vanemuise St., EE2400 Tartu, Estonia

E. López Períago, Department Edafoloxía e Química Agrícola, Fac. Farmacia, Campus Univ. S/N, 15706 Santiago de Compostela, Spain

R.L. Lowrance, USDA-Agricultural Research Service (ARS), Tifton, GA 31793, USA

C.P. Mainstone, Ecological Assessment Group, WRc plc, Medmenham, Henley Road, Marlow, UK

U. Mander, Institute of Geography, University of Tartu, 46 Vanemuise St., EE2400 Tartu, Estonia

P. Merot, Institut National de la Recherche Agronomique (INRA), Unité de Science du Sol et de Bioclimatologie, 65 route de saint Brieuc, 35042 Rennes Cedex, France

R.L. Mikkelsen, North Carolina State University, Department of Soil Science, Box 7619, Raleigh 27695-7619, USA

N.I. Ngapo, Environment Bay of Plenty, Whakatane, New Zealand

A. Núñez Delgado, Department Edafoloxía e Química Agrícola, Fac. Farmacia, Campus Univ. S/N, 15706 Santiago de Compostela, Spain

T.G. Parminter, AgResearch, Hamilton, New Zealand

J.E. Parsons, North Carolina State University, Department of Soil Science, Box 7619, Raleigh 27695-7619, USA

G. Pautou, UPR-ES 592 Laboratoire Ecosystèmes Alpins, Groupe Hydrosystèmes Alpins, Université Joseph Fourier, Centre de Biologie Alpine, B.P. 53, 38041 Grenoble (cedex 9), France

L. Patty, Insitut Technique des Céréales et des Fourrages, Lyon, France

C. Perala-Gardiner, Salix International Ltd, 18 Woodridge Close, Enfield, Middlesex, EN2 8HJ, UK

I. Perret, Ministry of Agriculture, Rural Areas and Forestry Directorate, Paris, France

G. Pinay, Laboratoire d'évolution des systéms naturels et modifiés, URA ECO Bio 1853, Campus de Beaulieu, Université de Rennes I, 35042 Rennes Cedex, France

D. Riddell-Black, Soil, Waste and Groundwater Group, WRc plc, Medmenham, Henley Road, Marlow, UK

B. Real, Insitut Technique des Céréales et des Fourrages (ITCF), Estrées-Mons, France

L. Ryszkowski, Research Centre for Agricultural and Forest Environments, Polish Academy of Science, ul. Bukowska 19, 60809 Poznan, Poland

G. Salmoiraghi, Department of Biology, University of Bologna, Via Selmi. 3, Bologna 40126, Italy

D.J. Schaeffer, Department of Veterinary Medicine/Biosciences, University of Illinois, Urbana, IL 61801, USA,

L.A. Schipper, Landcare Research NZ Ltd., Private Bag 3127, Hamilton, New Zealand

J.P. Schubauer-Berigan, Belle W. Baruch Institute for Marine Biology and Coastal Research, University of South Carolina, Columbia, SC 29208, USA and Winyah Bay-North Inlet National Research Reserve, PO Box 1630, Georgetown, SC 29442, USA

S.R. Smith, Soil, Waste and Groundwater Group, WRc plc, Medmenham, Henley Road, Marlow, UK

L. Stibe, County Administrative Board, S 301 86 Halmstad, Sweden

M.J. Stroud, National Institute of Water and Atmospheric Research, Hamilton, New Zealand

E. Turtula, Agricultural Research Centre of Finland, Institute of Crop and Soil Science, Finland

A. Tytherleigh, ADAS Statutory Centre, Staplake Mount, Starcross, Devon, EX6 8PE, UK

J. Uusi-Kämppä, Agricultural Research Centre of Finland, Institute of Soils and Environment, FIN – 31600, Jokoinen, Finland

T Yläranta, Agricultural Research Centre of Finland, Institute of Soils and Environment, FIN – 31600, Jokoinen, Finland

Introduction

1

N.E. Haycock[1], T.P. Burt[2], K.W.T. Goulding[3] and G. Pinay[4]

[1] *Quest Environmental, PO Box 45, Harpenden, Herts., AL5 5JL, UK*
[2] *Department of Geography, University of Durham, Science Laboratories, South Road, Durham, DH1 3LE, UK*
[3] *IARC Rothamsted Experimental Station, Harpenden, Herts., AL5 2JQ, UK*
[4] *Laboratoire d'évolution des systéms naturels et modifiés, URA ECO Bio 1853, Campus de Beaulieu, Université de Rennes I, 35042 Rennes Cedex, France*

INTRODUCTION

We decided to organise this conference for a number of reasons. Most importantly, there had not been an opportunity for all those engaged in research and implementation of buffer zones to meet formally and exchange ideas, problems and hopefully some solutions. Research into how and why buffer zones work was reaching a critical stage. Over 10 years of detailed research had been conducted (see Correll, this volume) and it was over 20 years since the merits of buffer zones for water quality were first discussed in western literature (e.g., Waikato Valley Authority, 1973). It is against this backdrop of scientific interest and a rising level of concern from land managers and government agencies that the conference was called. In the words of The Earl of Cranbrook, Chairman of English Nature and the opening speaker for the conference:

> *"English Nature believe that buffers must be an important element of any overall strategy… Perhaps the most important challenge, which we must not lose sight of, is to develop an overall strategy for land use. Such a strategy will identify in which landscapes buffer zones are best suited. Only then can we achieve the level of protection and enhancement of our river systems that is needed."*

We were overwhelmed by how positive the response was to the conference. All our first-choice speakers accepted the invitation to give papers and over 70 poster presentations were made, bringing together the best in the field. Overall, 21 nationalities were represented, a truly "international" conference.

The principal aim of the conference was to further our understanding of the value of buffer zone landscapes by examining the processes occurring within buffer zones in relation to their potential to conserve, enhance and protect the water environment. The use of natural buffer zones to protect freshwater from pollution has attracted considerable interest within the European Union and elsewhere. However, the factors accounting for the pollution retention capacity of buffer zones are diverse and, therefore, the performance of a buffer zone within a catchment is difficult to predict.

In recent years there has been a drive to integrate the results from research into management solutions that control diffuse pollution and protect the water environment. Some politically-driven initiatives have, thankfully, accelerated the debate on buffer zones but have, at the same time, stretched us to the limits of our knowledge on buffer zones. This drive for "solutions" has culminated in recent years in a number of major reviews. The first was a synthesis of Swedish research published in a special issue of Ambio (1994, volume 23, number 6) which documented the potential of a range of wetland habitats to trap nitrogen (also see Fleischer *et al.,* this volume). The second major review was the result of work within the Chesapeake Bay Program, Eastern USA (Lowrance *et al.*, 1995; Lowrance, this volume). Key summaries of the hydrogeological framework of riparian buffer zone investigations were developed as well as statements on the competence of riparian zones in these various hydrogeological settings. The third major publication, a combination of review paper and management document, was from the National Institute of Water and Atmospheric Research (NIWA), New Zealand (Collier *et al.*, 1995). This

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0

document included a synthesis of scientific data as well as management implications for farmers and agencies responsible for river and stream protection. The combination of scientific and practical advice should be central to any implementation policy and we were delighted when members of the New Zealand group accepted the invitation to present their work at the conference (Cooper, this volume; Downes *et al.,* this volume).

STRUCTURE OF THE BOOK

The proceedings consist of three parts: *Part I,* the processing of target chemicals within buffer zones; *Part II,* the potential of habitat types to act as buffer zones; *Part III,* the creation, restoration and long-term sustainability of buffer zones. We have drawn together in the book most of the spoken papers together with a small selection of the poster presentations. The titles of all the posters presented at the conference can be found in Appendix I.

Part I, the processing of target chemicals within buffer zones

This section seeks to review the key processes that account for the retention, transformation and ultimate fate of waterborne pollutants within buffer zones. Speakers were asked to focus on a key water quality parameter and review how this parameter is controlled within a range of habitats. Of key concern was the effect of buffer zone structure on its ability to function and what habitats are best for sediment, nitrogen, phosphorus or pesticide retention? Speakers were asked to comment on the optimal design or attributes required of a buffer zone, in order to achieve a particular function and also to comment on the limits of our current knowledge with suggestions of topics that need to be explored.

There were also special concerns about how the ecology and microbiology would react to nutrient inputs. Groffman focuses on the microbiological aspects and Dr. Chris Newbold (English Nature, Peterborough) spoke at the conference on the response of the vegetation complex (this paper is likely to be published elsewhere later this year).

Part II, the potential of habitat types to act as buffer zones

Here the focus is on specific habitats: in-stream, riparian, ponds, wetlands etc.; their potential to act as buffer zones is examined. Authors were asked to describe the principal features, hydrological regime and vegetation complexes of a particular habitat and comment on how these factors influence the dynamics of nutrient uptake. The authors were then asked to consider where in the catchment these habitats could be located to achieve their optimum potential and whether they need to be managed to sustain the key processes? Finally, we asked authors to discuss whether we know enough to make firm recommendations on how to preserve or restore these environments and, if so, what their outline suggestions would be for their chosen habitat.

The concluding papers in Part II develop similar themes but were asked to take a more global perspective; namely, to what extent can we model buffer zones (Gold and Kellogg) and the interactions between buffer zones and their catchment (Merot and Durand).

Part III, the creation, restoration and long-term sustainability of buffer zones

Part III sought to examine the strategies required to preserve or create buffer zones and the implications of buffer zone creation. The creation of buffer zones to control pollution is a change of land use which has a direct and multiple impact on the farmer. Dickson and Schaeffer and also Cook looked at the consequences of establishing buffer zones in the context of a farm. Tytherleigh discusses the lessons learnt from MAFF's Habitat Scheme Water Fringe Option, a pilot project which seeks to establish set-aside land along selected rivers in England. At the conference Dr. Terry Tooby (Pesticide Safety Directorate, UK) discussed in detail the problems of developing concepts of buffer zones in relation to the regulations on the use of pesticides. Tooby was unable to provide a written paper, but readers are directed to papers by Harris and Gril *et al.* (both in Part I) for a review of work on pesticide impacts and their fate within buffer zones. Downes *et al.* sought to examine the practical implications of restoring nitrogen buffers, while Addiscott was asked to take a critical look at whether buffer zones are a serious tool for the control of diffuse pollution. The final papers in this section sought to take a more strategic overview of the whole process. Gardiner and Perala-Gardiner discuss the role of integrated planning frameworks, particularly the role of catchment management plans, whilst Baudry looks at farm and landscape interactions. The section is concluded with a paper by Cooper who offers an insight into the New Zealand experience of encouraging implementation of buffer zones.

The editors have the final word and we present a concluding summary of the current concerns, as well as our own overview of what work needs to be done for the future development of the buffer zone concept. We believe there is now a sufficient basis in the literature to develop ideas and practices for buffer zone restoration. These proceedings give an insight into that foundation of knowledge.

REFERENCES

Ambio (1994) Special Edition: Wetlands as Nitrogen Traps. *Ambio*, **23**(6), 320-386.

Collier, K.J., Cooper, A.B., Davies-Colley, R.J., Rutherford, J.C., Smith, C.M. and Williamson, R.B. (1995) *Managing Riparian Zones: A contribution to protecting New Zealand's rivers and streams. Volumes I & II.* NIWA, Department of Conservation, Science Publications, P.O. Box 10, 420 Wellington, New Zealand.

Lowrance, R., Altier, L.S., Newbold, J.D., Schnabel, R.R., Groffman, P.M., Denver, J.M., Correll, D.L., Gilliam, J.W., Robinson, J.L., Brinsfield, R.B., Staver, K.W., Lucas, W.C. and Todd, A.H. (1995) *Water quality functions of riparian forest buffer systems in the Chesapeake Bay Watersheds.* U.S. Environmental Protection Agency, Chesapeake Bay Program Report. EPA 903-R-95-004.

Waikato Valley Authority (1973) *Lake Taupo – catchment control scheme. Final Report.* Published by WVA Hamilton, New Zealand.

Part I

The Processing of Target Chemicals Within Buffer Zones

Buffer zones and water quality protection: general principles

2

D.L. Correll

Smithsonian Environmental Research Center, P.O. Box 28, Edgewater, MD 21037, USA

Abstract
Riparian buffer zones (RBZ) improve water quality in different ways depending upon the pathway of delivery of water to the RBZ. Groundwater passing through the RBZ may be cleansed of nitrate and acidity due to a combination of denitrification, biostorage and changes in soil composition. Overland storm flows entering laterally from the uplands may be cleansed of suspended particulates, with adhering nutrients, inorganic toxins, and pesticides, as well as some dissolved nutrients and toxins. Sometimes these overland flows will also infiltrate within the RBZ and become a part of the groundwater, thus also obtaining the benefits associated with groundwaters in the RBZ. During stream flooding events, waters flooding out into the RBZ may also be cleansed of sediments, nutrients and toxic materials as a result of particulate trapping and the binding of materials on the leaf litter and soils within the RBZ. The RBZ is also an important source to the stream of high quality dissolved and particulate organic matter which is delivered both vertically and laterally. Forested RBZs also provide shade and evaporative cooling to streams, maintaining lower summertime temperatures critical to some biota. Factors which limit the effectiveness of the functions can be divided into internal and external. Factors external to the RBZ include watershed area and gradient, stream channel morphology, soil mineralogy and texture, bedrock type and depth, and climate. Factors internal to the RBZ include width and type of vegetation, waterlogging and organic content of soils, hydraulic conductivity, soil nutrient content and geochemistry. These water quality functions of RBZs and the factors which limit their effectiveness in various settings are reviewed from the world literature.

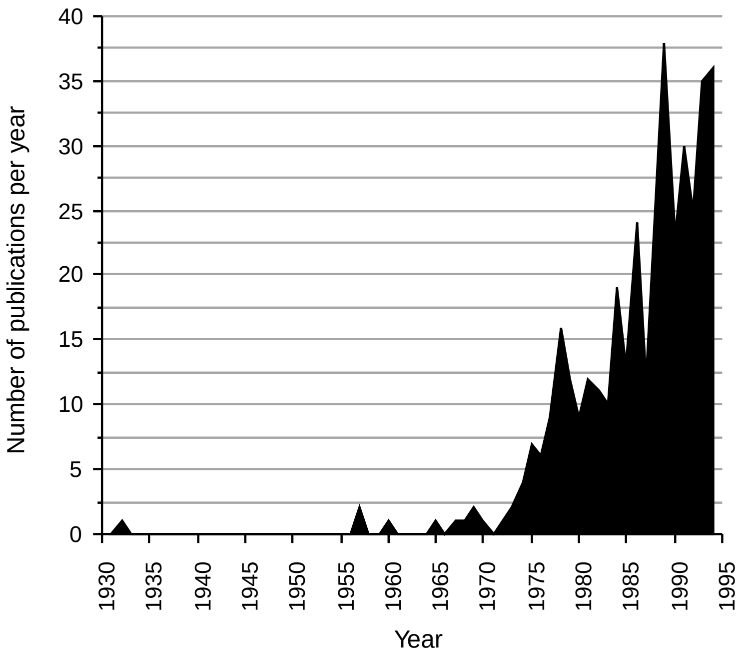
INTRODUCTION
Only a few papers which explicitly reported the results of studies of the water quality buffering effects of vegetated riparian zones were published prior to the early 1970s (Fig. 1). Although it is very difficult to draw sharp boundaries around this subject, there are now over 400 such papers and the rate of publication is about 30 to 35 papers per year. Despite this increasing rate of publication, our knowledge base concerning the water quality buffering effects of riparian zones is far from adequate. For descriptive aspects such as the removal of nitrate and particulate matter from the waters traversing the riparian zone or the release of organic matter to these waters we have considerable data but lack knowledge for how these functions change with climate or watershed management; species composition and age of the plant community; or how and when these functions will be saturated or exceeded. More fundamentally, we usually do not have an adequate understanding of the basic mechanisms involved in these processes or the controls over the rates of these processes. Lacking this understanding of mechanisms, we are in danger of being overconfident of our ability to predict system behaviour and response to major changes in inputs, invasions by exotic biota, or altered climate. With this situation in mind I will endeavour to review the world literature on riparian buffer zones, with my emphasis on streams and water quality.

If we look at riparian vegetation and the soils in which they grow as a part of the landscape (Lowrance *et al.*, 1985; Jordan *et al.*, 1986; Correll *et al.*, 1992), we can examine factors that control the health and functioning of these ecosystems and delimit them spatially. Some of these factors are internal or

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0

endogenous to the riparian ecosystems, while other factors are external or exogenous (associated with either the drainage basin or the stream channel) (Correll and Weller, 1989). Geomorphic factors may be internal (soil physics and chemistry, slope within the riparian zone) or external (watershed area and gradient, soil mineralogy and texture, bedrock type and depth, volume and composition of groundwater inputs, channel morphology). Watershed area and gradient are major determinants of the volume and kinetics of external inputs to riparian zones. Soil mineralogy is a very important determinant of the chemical composition of external inputs, whereas soil texture to a large extent determines the relative proportions of surface water and groundwater inputs. Lateral overland storm flows may be effectively cleaned by forested RBZs when these flows arrive as sheet flows from relatively small fields with slopes of 5% or less (Peterjohn and Correll, 1984), but when the fields are larger and storm flows become concentrated, they may erode channels through forested RBZs (Jordan *et al.*, 1993). Some success has been reported in treating these larger concentrated storm flows by the use of level spreaders and grassed filter strips prior to forested RBZs (Franklin *et al.*, 1992).

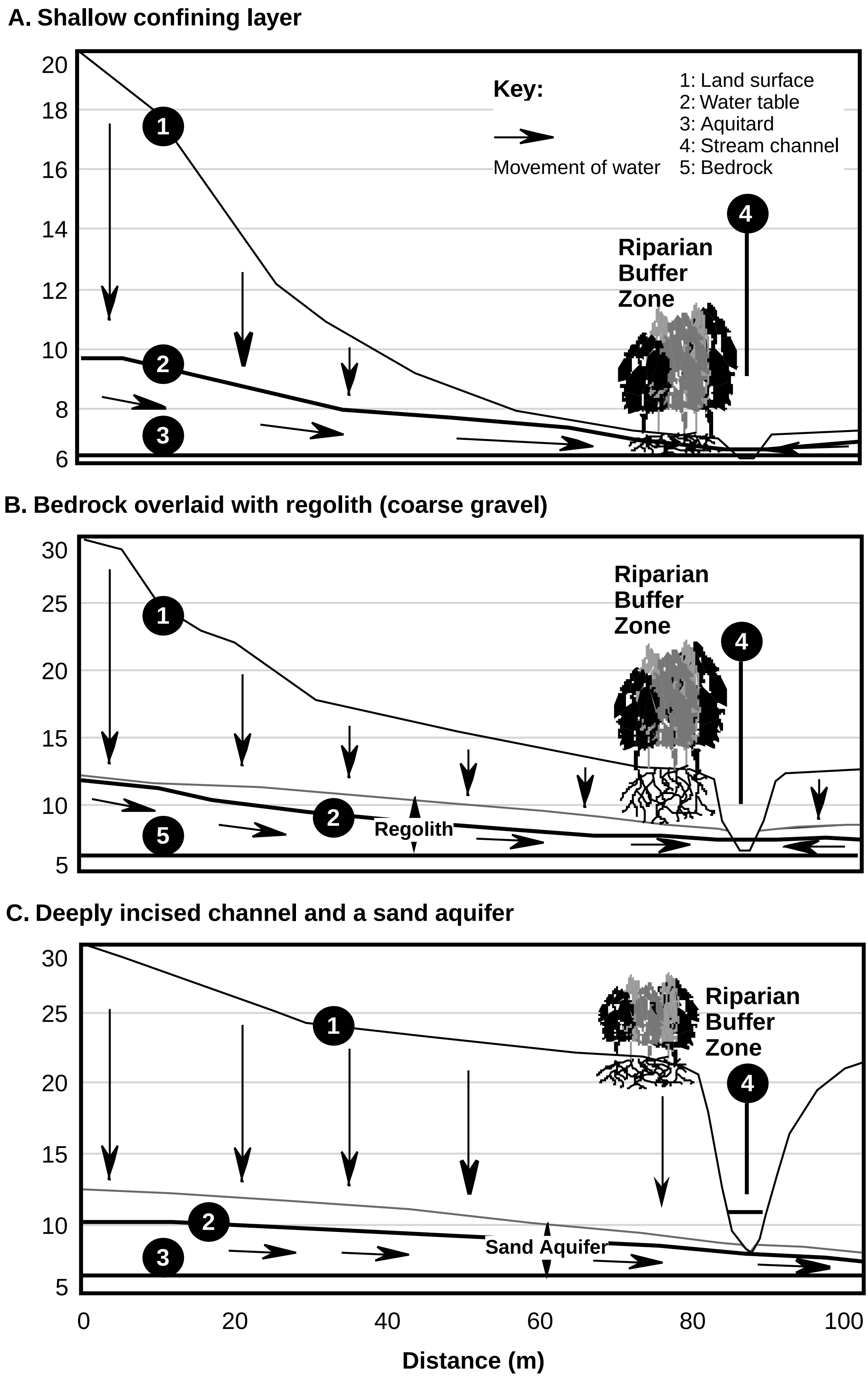
Figure 1. Number of publications related to riparian zones.



HYDROLOGY OF THE RIPARIAN BUFFER ZONE

Since the water quality effects of the Riparian Buffer Zone (RBZ) are highly dependent upon the volume and pathway of water movement through this zone, it is obvious that an understanding of hydrology is important (Fig. 2; Mitsch *et al.*, 1977; LaBaugh, 1986; Chescheir *et al.*, 1988; Correll and Weller, 1989; Dosskey and Bertsch, 1994). The major climatic control factors are the components of the hydrological cycle: precipitation, runoff, and evapotranspiration (ET) (Correll and Weller, 1989). ET is, in turn, governed primarily by such factors as vegetation, humidity, temperature, wind and sunlight. Thus, to some extent the riparian vegetation has a feedback to the hydrological cycle. The output of the external watershed equals precipitation minus ET minus infiltration to non-communicating deep aquifers. Depending upon the situation, some of this watershed output will be directly into the riparian zone from upslope areas and some of the output will be to the adjacent stream channel. Channel waters will have various amounts of inputs to the riparian zone via groundwater or surface flooding, depending upon stream discharge rate. The output of the riparian zone equals precipitation plus surface and groundwater inputs minus ET minus infiltration to deeper layers. If the local groundwater passes beneath the RBZ or the whole stream system at too great a depth, the riparian

Figure 2. Watershed cross-sections.



zone cannot interact (Fig. 2C; Denver, 1991; Staver and Brinsfield, 1991). In some cases overland storm flows entering the RBZ infiltrate the soils and become groundwater (Cooper *et al.*, 1995; Correll *et al.*, 1996). If this infiltration is not detected, data are easily misinterpreted. Stable isotope studies have shown that the source of water for ET of stream bank trees is sometimes lateral shallow groundwater flows rather than water from the channel (Dawson and Ehleringer, 1991). In a number of studies hydrological tracers such as bromide have been injected into the groundwater in order to demonstrate that sampling points downgradient were actually along the path of groundwater flow (Hill, 1990, 1991; Jordan *et al.*, 1993; Hubbard and Lowrance, 1994; Lockaby *et al.*, 1994). These tracer methods are usually qualitative rather than quantitative. In some cases (Triska *et al.*, 1989, 1990a ; Simmons *et al.*, 1992; Nelson *et al.*, 1995) the tracer was mixed with a nutrient such as nitrate or ammonium and was added to the system continuously for an extended time. Then ratios of nutrient to tracer were measured down gradient to detect nutrient transformations and to correct for the inevitable effects of dilution. An inadequate understanding of the hydrology of RBZ study sites usually limits the quantitative interpretation of study results.

OXIDATION / REDUCTION POTENTIAL OF THE SOILS

Many of the characteristics of riparian zones, such as the species composition of the vegetation and rates of processes such as denitrification, require that the soils be anaerobic or of low oxidation/reduction potential (Eh) at least part of the year. The vegetation of the riparian zone is of fundamental importance in maintaining this low Eh. The below-ground processes which result in this low Eh are composed of a series of biogeochemical reactions that occur in a defined order (Billen, 1976). These reactions transfer electrons from organic matter, released from the plants, to various terminal electron acceptors. The availability of terminal electron acceptors determines which level in the series will dominate below-ground processes at any one time and place in the riparian zone. Some of the more commonly important reactions are manganate ion reduction, denitrification, ferric iron reduction, sulphate reduction, and methanogenesis. They occur in this order as a result of thermodynamic considerations. None of these reactions can proceed in the presence of molecular oxygen. Once oxygen has been consumed by processes such as respiration, and sulphide and ammonium ion oxidation, then manganate reduction may proceed. Once all manganate is reduced or if none occurs at the site, then denitrification can proceed, and so on. The reversibility of many of the reactions is limited by the production of volatile end products or changing pH. These factors and others produce a series of negative feedback mechanisms which tend to limit the further progress of a given below-ground process (Correll and Weller, 1989). For example, in the case of sulphate reduction, as the ratio of electron acceptor to product decreases (e.g. $SO_4^{=}/S^{=}$), the equilibrium Eh required for the reaction to proceed declines. As the absolute concentration of sulphate declines, Eh must also decline for the reaction to proceed. As the pH rises, due to consumption of hydronium ions in the reaction, the Eh must decline for the reaction to proceed. At the same time the rates of entry of oxygen and other more easily reduced electron acceptors, such as nitrate, continue at previous rates, which will raise the Eh if sulphate reduction rates begin to slow down. Another example is denitrification. As the reaction proceeds, pH rises due to hydronium ion consumption in the reaction, and nitrate is converted to dinitrogen and nitrous oxide gases which evolve from the system, and the rate of denitrification slows while the rates of other processes such as nitrification may increase. For the riparian zone to maintain a low Eh it is therefore essential in the long run that the plants have a high primary productivity and that enough of the resulting photosynthate be released below-ground to provide enough electrons to drive these reactions at high rates. Despite the relative ease of measuring soil Eh, few studies have reported this critical parameter. Exceptions include Jacobs and Gilliam (1983, 1985), Davidson and Swank (1986), Pinay *et al.* (1989), Jordan *et al.* (1993) and Correll *et al.* (1996).

RIPARIAN EFFECTS ON THE GROUNDWATER ENTERING THE STREAM

Most of the water flowing down the channel of most streams reached the channel at some point as groundwater moving from a recharge area to the stream. This groundwater may move as shallow lateral flows (Fig. 2A) or as deeper flows that surface in the channel from strata below the channel (Fig. 2B). Of course a little of the rain water is deposited directly in the channel (channel interception) and some arrives as overland storm flows. The overland storm flows are of short duration following intense rain events and are usually less than 30% of the annual stream discharge, with the exception of systems with very heavy clay soils where infiltration is very limited. Thus, the quality of the water in the channel during baseflow periods between major storms is highly dependent upon processes within the RBZ. As used here the RBZ includes much of what is sometimes referred to as the hyporheic zone.

Nitrogen transformations

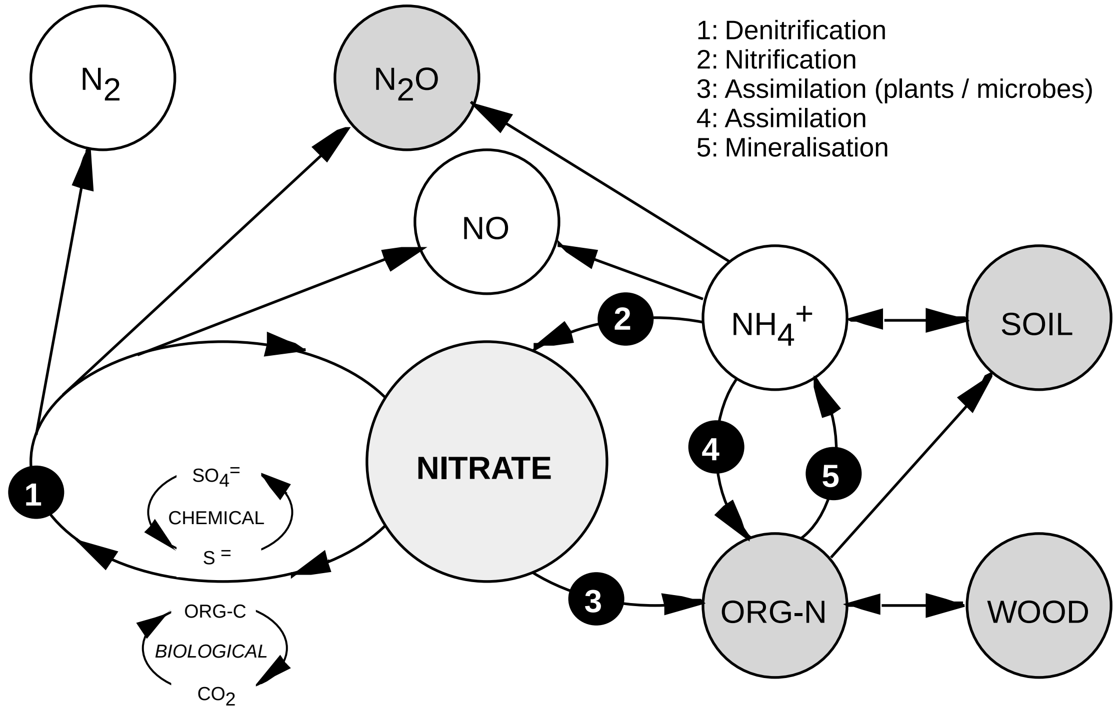
The first studies which directly measured nitrate concentration decreases in groundwater as it moved through riparian zones along streams were in the Coastal Plain of North Carolina (Gilliam *et al.*, 1974; Gambrell *et al.*, 1975). High concentrations of nitrate in shallow groundwater percolating out of row crop fields declined rapidly before reaching the stream channels and streams in these areas did not have the high nitrate concentrations one would expect from the watershed land use. In the early 1980s these results were repeated and more directly related to the presence of deciduous hardwood forests in the RBZs at three Atlantic Coastal Plain sites: Little River watershed near Tifton, Georgia (Lowrance *et al.*, 1983; Lowrance *et al.* 1984a, 1984b, 1984c), two sites in North Carolina (Jacobs and Gilliam, 1983, 1985), and the Rhode River watershed in Maryland (Correll, 1983; Correll *et al.*, 1984; Peterjohn and Correll, 1984). All of these sites were in regions of the Coastal Plain where most of the discharge from agricultural fields moves as shallow lateral groundwater flows to stream channels. Thus, it moves through the rooting zone of riparian forests in soils that are often waterlogged and maintain a low oxidation/reduction potential. These studies led to mass balances for total nitrogen retention of 74 kg N/ha yr or 89% of inputs at the Rhode River site and 26 kg N/ha yr or 67% of inputs at the Little River site. At the Beaverdam Creek site in North Carolina nitrate retention was 30 kg N/ha or 85% of nitrate inputs.

These early studies were of great interest to both environmental scientists and land managers. Within a few years similar studies were publishing results from a series of new and different sites. These include the Garonne River and its tributaries in France (Pinay and Décamps, 1988; Pinay *et al.* 1989); intensively managed sheep pastures in New Zealand with grassed RBZs (Cooke and Cooper, 1988; Cooper, 1990; Schipper *et al.*, 1994a, 1994b); grassed and forested RBZs in England (Haycock and Burt, 1993a, 1993b; Haycock and Pinay, 1993); forested RBZs in Rhode Island (Groffman *et al.*, 1991, 1992; Hanson *et al.*, 1994); the Mahantango Creek watershed in the Ridge and Valley area of Pennsylvania (Schnabel, 1986; Schnabel and Stout, 1994); forested and grassed RBZs in Illinois (Osborne and Kovacic, 1993); Ontario, Canada (Warwick and Hill, 1988), and two additional sites in the Atlantic Coastal Plain on the eastern shore of Maryland (Jordan *et al.*, 1993; Correll *et al.*, 1996). At all of these sites lateral flows of groundwater had decreases in nitrate concentration as they moved through the RBZ. More limited studies were also reported from Germany (Knauer and Mander, 1989) and Estonia (Mander *et al.*, 1989).

Mechanisms of nitrate removal or transformation in the RBZ

The mechanisms responsible for these widely documented retentions of nitrate have proven rather elusive. Candidate mechanisms include denitrification, assimilation and retention by the vegetation, and transformation to ammonium and organic nitrogen followed by retention in the soils of the RBZ (Fig. 3). It is quite clear in a number of studies that the nitrate is not simply converted to other soluble forms of nitrogen and discharged in the stream channel (e.g. Lowrance *et al.* 1983, 1984a, 1984b, 1984c ; Peterjohn and Correll, 1984; Correll and Weller, 1989; Jordan *et al.*, 1993; Correll *et al.*, 1996). Few studies have accurately measured the amount of nitrate removed by any one of these mechanisms at a given site and no study has measured the removal rate by all three mechanisms.

Figure 3. Mechanisms of nitrate removal in riparian buffer zones.



Denitrification is most often invoked as the primary mechanism of nitrate retention; however the extreme spatial and temporal variability of denitrification rates in RBZs make it very difficult to determine accurate fluxes (Correll, 1991; Weller *et al.,* 1994). Laboratory incubations of soil samples demonstrate the potential for denitrification, but these measurements are hard to extrapolate to the field. The products of denitrification include dinitrogen, nitrous oxide and nitric oxide and the proportions of these products are highly variable. Most studies do not measure all of these products. Instead they use chemical inhibitors to stop the production of dinitrogen. These inhibitors may cause more artefacts.

Field studies of nitrate mass balance show that nitrate is effectively removed: a) at all times of the year in temperate climates; and b) from groundwaters moving in subsoils at depths of several metres. Studies of potential denitrification in riparian soils find that most of this potential is in the top few centimetres of soil (Ambus and Lowrance, 1991; Ambus, 1993; Pinay *et al.*, 1993). Conditions in the deeper subsoil include low temperature, low pH and low concentrations of dissolved organic matter. These facts lead some to scientists to conclude that assimilation by the vegetation is the primary mechanism of nitrate removal (e.g. Fail *et al.*, 1986). While the vegetation may be very important in explaining the nitrate removal, nitrate is removed in the winter at sites where the vegetation is hardwood deciduous forest that is dormant in the winter. Further, measurements of nitrogen accumulation in the annual accretion of woody biomass was only 12 to 20 kg N/ha yr (Peterjohn and Correll, 1984; Correll and Weller, 1989). Thus, even if all of the nitrogen for this biomass came from nitrate in the groundwater it would only account for about 30% of the nitrate removal. It is highly likely that some of the nitrogen assimilation occurred from other sources. Jacobs and Gilliam (1983) also found that accumulation of nitrogen in riparian vegetation explained a relatively minor portion of the nitrate that was removed at their sites. These facts make it likely that assimilation and storage in woody biomass is a significant mechanism, but not the primary mechanism. However, it is possible that assimilation by the vegetation and recycling to the forest floor as litter is important in unravelling the overall primary mechanism. The rate of assimilation at the Rhode River site was 77 kg N/ha yr and litter fall plus throughfall was 66 kg N/ha yr (Peterjohn and Correll, 1984). Thus, assimilation by the forest could be the primary mechanism of nitrate removal from groundwater during the growing season and the flux of organic nitrogen delivered to the forest floor as litter could be gradually mineralised and denitrified at the soil surface. Such a mechanism would, however, not explain nitrate removal during the winter. Studies in New Zealand have shown high rates of dissimilative nitrate reduction to ammonium, which is then bound to the soils of the RBZ (Schipper *et al.,* 1994a). Some scientists believe that nitrate removal from the groundwater traversing RBZ subsoils is accomplished by chemical rather than biological denitrification (Mariotti *et al.,* 1988). Strong reducing agents such as iron sulphides may react with nitrate to produce dinitrogen and sulphate. Certainly some RBZs at times reduce sulphate to sulphide (Correll and Weller, 1989; Jordan *et al.,* 1993), and at times sulphide is oxidised back to sulphate and released (Correll and Weller, 1989). If chemical denitrification does take place, it still would depend, in the long–term, upon adequate supplies of organic matter in the subsoils of the RBZ to maintain a low enough Eh. Understanding the primary mechanism of nitrate removal is very important. If this is the primary mechanism at such sites, the system's soils will eventually become nitrogen saturated. Some evidence of potential nitrogen saturation has been demonstrated at sites in Rhode Island (Hanson *et al.,* 1994). If RBZs do become nitrogen saturated and cease to carry out their nitrate removal function the result would be very serious declines in receiving water quality.

In some RBZs soil Eh is not low enough some of the time to allow denitrification at significant rates. This may be short–term, as associated with periods between rainfall events, or longer-term due to extended drought, or it may be a seasonal phenomenon. It may also be spatially very patchy (e.g. Weller *et al.,* 1994). In such situations an alternation of denitrification and nitrification are observed (e.g. Duff and Triska, 1990; Triska *et al.,* 1990b, 1993; Jones *et al.,* 1994). In both processes significant nitrogen losses as nitrous oxide and nitric oxide occur in addition to the production of dinitrogen by denitrification. Some researchers believe that this alternation of nitrification and denitrification help to explain the large nitrate removals observed in RBZs.

Other effects on groundwater quality

Although RBZ effects on nitrogen, especially nitrate, are often emphasised in studies of riparian buffering, there are other significant water quality effects. The pH of groundwater is often significantly altered. Below-ground processes often consume or release hydronium ions (Correll and Weller, 1989). In non-calcareous, poorly buffered soils, such as are found on parts of the Atlantic Coastal Plain, groundwater draining from row crops is quite acidic, due to the effects of nitrification in the fields and acid deposition (Correll *et al.* 1987). As it moves through RBZs both plant assimilation of nitrate and denitrification consume hydronium ions and the pH increases to values less toxic to aquatic animals (Peterjohn and Correll, 1986; Jordan *et al.,* 1993). These pH changes, when coupled to other data, can be used to calculate metabolic rates (e.g. Jones Jr. *et al.,* 1994). Dissolved phosphorus and organic carbon concentrations usually increase as groundwater moves through RBZs (Correll and Weller, 1989; Jordan *et al.,* 1993). This is the result of the low Eh in these riparian soils.

SEDIMENT, NUTRIENT AND PESTICIDE TRAPPING ON THE SURFACE

The role of riparian vegetation in trapping sediments and adhering phosphorus was reported in widely cited articles by Karr and Schlosser (1978) and Schlosser and Karr (1981a, b). Riparian Vegetation facilitates the removal of suspended sediments, along with their nutrient contents, from two types of surface water: a) overland storm water entering laterally (Mitsch *et al.,* 1979; Peterjohn and Correll, 1984; Lowrance *et al.,* 1988; Klarer and Millie, 1989; Chescheir *et al.,* 1991; Parsons *et al.,* 1994); and b) flood waters entering from the stream channel (Kitchens *et al.,* 1975; Hart *et al.,* 1987; Kleiss et al., 1989; Hupp and Morris 1990; Hupp *et al.,* 1993; Johnston, 1993; Brunet *et al.,* 1994). In both cases riparian vegetation plays an important role in removing and retaining particulates. Increased friction with soil surfaces can cause reduced velocity and consequent sedimentation of particulates, but riparian vegetation and the layer of litter it deposits on the soil surface are much more effective at slowing the velocity of the surface waters. The fine roots of the plants, which are concentrated on or near the surface, and the microbial communities on the surfaces of the soil, litter, and above-ground plant organs also are able to assimilate dissolved nutrients from the surface waters (Peterjohn and Correll, 1984). Relatively few studies have examined the effectiveness of RBZs in trapping pesticides and other toxic materials from overland storm flows. Early work by Correll *et al.* (1978); Rhode *et al.* (1980), and Asmussen *et al.* (1977) found that most of the herbicides atrazine, alachlor, trifluralin, and 2,4-D were removed from cropland discharges. More recently Schultz *et al.* (1994) found that a reconstructed three-tiered riparian buffer in Iowa was effective in removing atrazine from cropland discharges.

IS THE VEGETATION IMPORTANT OR NECESSARY?

Questions are often asked about the efficacy or necessity for vegetation in the RBZ. Does it matter whether vegetation is present? Is woody vegetation more effective than grass or herbaceous vegetation? Are broad-leaved hardwoods better than coniferous trees? How wide a zone of vegetation is needed? These are all good questions for which we have few good answers. Although there was general agreement among studies of riparian forest buffers in the Atlantic Coastal Plain that nitrate was efficiently removed from shallow groundwater (Jacobs and Gilliam, 1983; Lowrance *et al.* 1983; Peterjohn and Correll, 1984), this in itself did not prove that the forest was necessary for this process. Studies on the North Carolina Coastal Plain found that fields could be cropped right up to the stream channel and nitrate removal would still occur efficiently, if controlled drainage structures were used to prevent the drying of the riparian soils (Gilliam *et al.*, 1979, 1986). Groffman *et al.* (1991) reported that denitrification potentials in surface soils of grassed RBZs in Rhode Island were somewhat higher than in forested RBZs. Further, Haycock and Burt (1993a, 1993b) found that grass riparian zones in England were very effective in nitrate removal from groundwater. Haycock and Pinay (1993) found that poplar forested RBZs were somewhat more effective at nitrate removal than grass, especially in the winter. Osborne and Kovacic (1993) found that forested RBZs in Illinois were more effective than grass for nitrate removal, but less effective for removal of phosphate and dissolved organic phosphorus from groundwater. Finally, Correll *et al.* (1996), in a comparison of two adjacent RBZ sites in Maryland, one grassed and one forested, found that they had similar nitrate removal efficiencies.

While there is considerable uncertainty on the exact role of riparian vegetation and the relative efficacy of various types of vegetation, it seems clear that grass or dense herbaceous vegetation is more effective at trapping particulates from overland storm flows (Osborne and Kovacic, 1993; Parsons *et al.*, 1994), but that woody vegetation may be more effective at removing nitrate from groundwater. In the long term it is clear that riparian vegetation is necessary to maintain the organic matter in riparian soils, which is needed for maintaining low Eh and processes such as denitrification. Woody vegetation, especially forest, is also more effective at providing organic matter in the deeper subsoils, where it is needed for effective denitrification in groundwater. However, changes in soil chemistry are slow and therefore the effects of present land management may not be apparent for decades.

RIPARIAN VEGETATION AS A SOURCE OF ORGANIC MATTER TO THE CHANNEL

Most stream channels are partially heterotrophic ecosystems which rely on organic matter inputs from the riparian zone and watershed for much of the organic matter used as an energy source to drive their food webs (Fisher and Likens, 1973; Minshall, 1978; Triska *et al.*, 1982; Conners and Naiman, 1984; Cuffney, 1988; Kleiss *et al.*, 1989). Many studies have shown that most of the particulate organic matter and much of the dissolved organic matter inputs are derived from areas immediately adjacent to the stream channel (Sedell *et al.*, 1974; McDowell and Fisher, 1976; Winterbourn, 1976; Triska *et al.*, 1984; Sidle, 1986; King *et al.*, 1987; Chauvet and Jean-Louis, 1988; Cushing, 1988; Gurtz *et al.*, 1988; Benson and Pearson, 1993; Sweeney, 1993). Vegetation along the stream bank and overhanging the channel contributes vertical litter fall and vegetation near the bank contributes litter by downslope lateral movement. The relative proportion of these two types of input varies dependent upon the local situation.

Role of riparian vegetation in stream temperature control
Riparian forests reduce solar heating of stream water by shading, especially in low order streams (Brown and Krygier, 1970). Any riparian vegetation provides cooling by evapotranspiration of soil water and shallow groundwater (Beschta, 1984; Theuer *et al.*, 1984; Sinokrot and Stefan, 1993). This cools waters flowing into the stream channel laterally and waters exchanging between the channel and the hyporheic zone. The evapotranspiration cooling is greatest when the vegetation is forest, since forest has the highest leaf area index and consequently the highest evapotranspiration rates. Hardwood deciduous riparian forest in temperate climates has evapotranspiration rates as high as 118 cm per year (Peterjohn and Correll, 1986).

Riparian vegetation as a source of large woody debris
Stream channels benefit from a steady input of woody branches and tree trunks. This debris and the debris dams that result bring about complexity in channel morphology and more useful habitats for stream biota (Minshall, 1978). The necessary woody debris originates almost entirely in the riparian zone (Webster, 1977). This literature will not be reviewed here; for a comprehensive review see Harmon *et al.* (1986).

RIPARIAN BUFFERS AND FORESTRY

When forested watersheds are clear-cut the rates of erosion and nutrient leaching increase until the vegetation begins to recover. Studies of the effects of clear-cuts on stream water quality have been reviewed by Vitousek (1981) and Webster *et al.* (1992) and will not be discussed here. However, in a few studies paired forested watersheds were subjected to experimental logging. Typically, the streams draining all would be monitored for several years, then one or more would be clear-cut, some would be clear-cut but a forest buffer strip would be retained, and at least one would be maintained as a forested control. Following the logging the water quality of the streams would be monitored to document the speed of recovery. Examples of such studies include West Virginia (Aubertin and Patric, 1974), Pennsylvania (Lynch and Corbett, 1990); and western Australia (Borg *et al.*, 1988); the results were similar. If a forested buffer was maintained, only small increases in suspended sediments and nutrients were observed in the streams compared with controls and complete clear-cuts.

RESTORATION OF FORESTED RIPARIAN BUFFERS

Once land managers became aware of the benefits to be gained from RBZs, a movement began to restore these vegetated buffers in areas where they had been destroyed. Two research projects in the United States have been attempting to re-establish native hardwood forest in stream riparian zones while monitoring their effectiveness. Several projects are under way on the Little River watershed in Georgia and early results indicate some success both in establishing forest and intercepting agricultural pollution (Hubbard *et al.*, 1995; Lowrance *et al.*, 1995). Another project on the White Clay Creek watershed in Pennsylvania encountered more difficulty in re-establishing hardwood forest, but is now making progress (Sweeney, 1993). In some cases the sapling trees need protection from browsing and girdling as well as intense competition from exotic plants before they can become re-established.

SUMMARY

The efficacy of RBZs in removing pollutants from surface and groundwater is highly dependent upon hydrology. For effective removal of particulates and dissolved nutrients and toxic materials, surface flows must occur as sheet flow rather than highly focused flows. For effective removal of nitrate and acidity, groundwater must move through the RBZ at a slow speed and at a shallow enough depth to be within the rooting zone of riparian vegetation. The vegetation in the RBZ must provide enough friction to surface flows to improve the efficiency of particulate trapping and provide surface litter to facilitate the assimilation of dissolved nutrients and toxic materials. The vegetation must also release enough organic matter at the depth of the groundwater to maintain a low enough Eh to allow rapid rates of denitrification. Riparian vegetation also provides shading and evaporative cooling of the stream channel. Cooling is most effective if the vegetation has a high leaf area index which reaches a maximum in hardwood deciduous forest. Vegetation also provides the stream channel communities with litter and large woody debris.

REFERENCES

Ambus, P. (1993) Control of denitrification enzyme activity in a streamside soil. *FEMS Microbial Ecol.*, **102**, 225-234.

Ambus, P. and Lowrance, R. (1991) Comparison of denitrification in two riparian soils. *Soil Sci. Soc. Am. J.*, **55**, 994-997.

Asmussen, L.E., White, A.W., Hansen, E.W. and Sheridan, J.M. (1977) Reduction in 2,4-D load in surface runoff down a grassed waterway. *J. Environ. Qual.*, **6**, 159-162.

Aubertin, G.M. and Patric, J.H. (1974) Water quality after clear cutting a small watershed in West Virginia. *J. Environ. Qual.*, **3**, 243-249.

Benson, L.J. and Pearson, R.G. (1993) Litter inputs to a tropical Australian rainforest stream. *Aust. J. Ecol.*, **18**, 377-383.

Beschta, R.L. (1984) TEM84: *A computer model for predicting stream temperatures resulting from the management of streamside vegetation.* Report WSDG-AD-00009, USDA ForestService, Watershed Systems Development Group, Fort Collins, CO, 76 pp.

Billen, G. (1976) The dependence of the various kinds of microbial metabolism on the redox state of the medium. In: *Biogeochemistry of Estuarine Sediments,* UNESCO, Paris, France, pp. 254-326.

Borg, H., Hordacre, A. and Batini, F. (1988) Effects of logging in stream and river buffers on watercourses and water quality in the southern forest of Western Australia. *Australian For.*, **51**, 98-105.

Brown, G.W. and Krygier, J.T. (1970) Effects of clear-cutting on stream temperature. *Water Resources Res.*, **6**, 1133-1139.

Brunet, R.C., Pinay, G., Gazelle, F. and Roques, L. (1994) Role of the floodplain and riparian zone in suspended matter and nitrogen retention in the Adour River, south-west France. Regulated Rivers; *Research and Management,* **9**, 55-63.

Chauvet, E. and Jean-Louis, A.M. (1988) Production de litiere de la ripisylve de la Garonne et apport au fleuve. *Acta Oecologia, Oecologia Generalis,* **9**, 265-279.

Chescheir, C.M., Skaggs, R.W., Gilliam, J.W. and Broadhead, R.G. (1988) Hydrology of wetland buffer areas for pumped agricultural drainage water. In: Hook, D.D. (Ed.) *The Ecology and Management of Wetlands.* Timber Press, Portland, OR, pp. 260-274.

Chescheir, C.M., Gilliam, J.W. Skaggs, R.W. and Broadhead, R.G. (1991) Nutrient and sediment removal in forested wetlands receiving pumped agricultural drainage water. *Wetlands,* **11**, 87-103.

Cooke, J.G. and Cooper, A.B. (1988) Sources and sinks of nutrients in a New Zealand catchment. III. Nitrogen. *Hydrol. Proc.*, **2**, 135-149.

Cooper, A.B. (1990) Nitrate depletion in the riparian zone and stream channel of a small headwater catchment. *Hydrobiologia,* **202**, 13-26.

Cooper, A.B., Smith, C.M. and Smith, M.J. (1995) Effects of riparian set-aside on soil characteristics in an agricultural landscape: Implications for nutrient transport and retention. *Agric. Ecosyst. Environ.*, **55**, 61-67.

Conners, M.E. and Naiman, R.J. (1984) Particulate allochthonous inputs: relationships with stream size in an undisturbed watershed. *Canad. J. Fish. Aquat. Sci.*, **41**, 1473-1488.

Correll, D.L., Pierce, J.W. and Wu, T.L. (1978) Studies of the transport of atrazine and alachlor from minimum till corn fields into Chesapeake Bay tidal waters. *Proc. Northeast. Weed Sci. Soc.*, **32**(sup.), 21-32.

Correll, D.L. (1983) N and P in soils and runoff of three coastal plain land uses. In: Lowrance, R., Assmussen, L. and Leonard R. (Eds.) *Nutrient Cycling in Agricultural Ecosystems.* Univ. Georgia Press, Athens, GA, pp. 207-224.

Correll, D.L. (1991) Human impact on the functioning of landscape boundaries. In: Holland, M.M., Risser, P.G. and Naiman, R.J. (Eds.) *The Role of Landscape Boundaries in the Management and Restoration of Changing Environments.* Chapman and Hall, New York, pp. 90-109.

Correll, D.L., Goff, N.M. and Peterjohn, W.T. (1984) Ion balances between precipitation inputs and Rhode River watershed discharges. Bricker, O. (Ed.) *Geological Aspects of Acid Deposition.* Ann Arbor Science, Ann Arbor, MI, pp. 77-111.

Correll, D.L., Miklas, J.J., Hines, A.H. and Schafer, J.J. (1987) Chemical and biological trends associated with acidic atmospheric deposition in the Rhode River watershed and estuary. *Water Air Soil Pollut.*, **35**, 63-86.

Correll, D.L. and Weller, D.E. (1989) Factors limiting processes in freshwater wetlands: an agricultural primary stream riparian forest. In: Sharitz, R.R. and Gibbons, J.W. (Eds.) *Freshwater Wetlands and Wildlife.* USDOE, Oak Ridge, TN, USA, pp. 9-23.

Correll, D.L., Jordan, T.E. and Weller, D.E. (1992) Nutrient flux in a landscape: Effects of coastal land use and terrestrial community mosaic on nutrient transport to coastal waters. *Estuaries,* **15**, 431-442.

Correll, D.L., Jordan, T.E. and Weller, D.E. (1996) Failure of agricultural riparian buffers to protect surface waters from groundwater contamination. In: Gibert, J. (Ed.) *Groundwater-Surface Water Ecotones.* Cambridge Univ. Press, London, in press.

Cuffney, T.F. (1988) Input, movement and exchange of organic matter within a subtropical coastal blackwater river-floodplain system. *Freshwater Biol.*, **19**, 305-320.

Cushing, C.E. (1988) Allochthonous detritus input to a small, cold desert spring-stream. *Verh. Internatl. Theoret. Angewan. Limn.*, **23**, 1107-1113.

Davidson, E.A. and Swank, W.R. (1986) Environmental parameters regulating gaseous nitrogen losses from two forested ecosystems via nitrification and denitrification. *Applied Environ. Microbiol.*, **52**, 1287-1292.

Dawson, T.E. and Ehleringer, J.R. (1991) Streamside trees that do not use stream water. *Nature,* **350**, 335-337.

Denver, J.M. (1991) Groundwater-sampling network to study agrochemical effects on water quality in the unconfined aquifer. In: *Groundwater Residue Sampling Design.* Amer. Chem. Soc. Symp. #465, pp. 139-149.

Dosskey, M.G. and Bertsch, P.M. (1994) Forest sources and pathways of organic matter transport to a blackwater stream: a hydrologic approach. *Biogeochemistry,* **24**, 1-19.

Duff, J.H. and Triska, F.J. (1990) Denitrification in sediments from the hyporheic zone adjacent to a small forested stream. *Can. J. Fish. Aquatic Sci.*, **47**, 1140-1147.

Fail, J.L., Hamzah, M.N., Haines, B.L. and Todd, R.L. (1986) Above and below ground biomass, production, and element accumulation in riparian forests of an agricultural watershed. In: Correll, D.L. (Ed.), *Watershed Research Perspectives.* Smithsonian Press, Wash. DC, pp. 193-224.

Fisher, S.G. and Likens, G.E. (1973) Energy flow in Bear Brook, New Hampshire: an integrative approach to stream ecosystem metabolism. *Ecol. Monogr.*, **43**, 421-439.

Franklin, E.C., Gregory, J.D. and Smolen, M.D. (1992) *Enhancement of the Effectiveness of Forested Filter Zones by Dispersion of Agricultural Runoff.* Report No. UNC-WRRI-92-270, Water Resources Res. Inst., Raleigh, NC, 28 pp.

Gambrell, R.P., Gilliam, J.W. and Weed, S.B. (1975) Denitrification in subsoils of the North Carolina coastal plain as affected by soil drainage. *J. Environ. Qual.*, **4**, 311-316.

Gilliam, J.W., Daniels, R.B. and Lutz, J.F. (1974) Nitrogen content of shallow ground water in the North Carolina coastal plain. *J. Environ. Qual.*, **3**,147-151.

Gilliam, J.W., Skaggs, R.W. and Weed, S.B. (1979) Drainage control to diminish nitrate loss from agricultural fields. *J. Environ. Qual.*, **8**, 137-142.

Gilliam, J.W., Skaggs, R.W. and Doty, C.W. (1986) Controlled agricultural drainage: An alternative to riparian vegetation. In: Correll, D.L. (Ed.), *Watershed Research Perspectives,* Smithsonian Press, Wash. DC, pp. 225-243.

Groffman, P.M., Axelrod, E.A., Lemunyon, J.L. and Sullivan, W.M. (1991) Denitrification in grass and forest vegetated filter strips. *J. Environ. Qual.*, **20**, 671-674.

Groffman, P.M., Gold, A.J. and Simmons, R.C. (1992) Nitrate dynamics in riparian forests: Microbial studies. *J. Environ. Qual.*, **21**, 666-671.

Gurtz, M.E., Marzolf, G.R., Killingbeck, K.T., Smith, D.L. and McArthur, J.V. (1988) Hydrologic and riparian influences on the import and storage of coarse particulate organic matter in a prairie stream. *Can. J. Fish. Aquat. Sci.*, **45**, 655-665.

Hanson, G.C., Groffman, P.M. and Gold, A.J. (1994) Symptoms of nitrogen saturation in a riparian wetland. *Ecol. Appl.*, **4**, 750-756.

Harmon, M.E., Franklin, J.F., Swanson, F.J., Sollins, P., Gregory, S.V., Lattin, J.D and Anderson, N.H. (1986) Ecology of coarse woody debris in temperate ecosystems. *Adv. Ecol. Res.*, **5**, 133-302.

Hart, B.T., Ottaway, E.M. and Noller, B.N. (1987) Magela Creek system, northern Australia. II. Material budget for the floodplain. *Aust. J. Mar. Freshwater Res.*, **38**, 861-876.

Haycock, N.E. and Burt, T.P. (1993a) The sensitivity of rivers to nitrate leaching: The effectiveness of near-stream land as a nutrient retention zone. In: Thomas, D.S.G. and Allison, R.J. (Eds.), *Landscape Sensitivity,* Wiley, London, pp. 261-272.

Haycock, N.E. and Burt, T.P. (1993b) Role of floodplain sediments in reducing the nitrate concentration of subsurface runoff: a case study in the Cotswolds, UK. *Hydrol. Processes.*, **7**, 287-295.

Haycock, N.E. and Pinay, G. (1993) Groundwater nitrate dynamics in grass and poplar vegetated riparian buffer strips during the winter. *J. Environ. Qual.*, **22**, 273-278.

Hill, A.R. (1990) Ground water flow paths in relation to nitrogen chemistry in the near-stream zone. *Hydrobiologia,* **206**, 39-52.

Hill, A.R. (1991) A ground water nitrogen budget for a headwater swamp in an area of permanent ground water discharge. *Biogeochemistry*, **14**, 209-224.

Hubbard, R.K. and Lowrance, R.R. (1994) Spatial and temporal patterns of solute transport through a riparian forest. In: *Riparian Ecosystems in the Humid U.S., Functions, Values and Management*, Natl. Assoc. Conserv. Districts, Wash. D.C., pp. 403-411.

Hubbard, R.K., Vellidis, G., Lowrance, R., Davis, J.G. and Newton, G.L. (1995) Using riparian buffers to treat animal waste. In: Steele, K. (Ed.), *Animal Waste and the Land-Water Interface*, Lewis, New York, pp. 127-134.

Hupp, C.R. and Morris, E.E. (1990) A dendrogeomorphic approach to measurement of sedimentation in a forested wetland, Black Swamp, Arkansas. *Wetlands*, **10**, 107-124.

Hupp, C.R., Woodside, M.D. and Yanosky, T.M. (1993) Sediment and trace element trapping in a forest wetland, Chickahominy River, Virginia. *Wetlands*, **13**, 95-104.

Jacobs, T.C. and Gilliam, J.W. (1983) *Nitrate loss from agricultural drainage waters: Implications for nonpoint source control.* University of North Carolina, Raleigh, NC. 208 pp.

Johnston, C.A. (1993) Material fluxes across wetland ecotones in northern landscapes. *Ecol. Appl.*, **3**, 424-440.

Jones Jr., J.B., Holmes, R.M., Fisher, S.G. and Grimm, N.B. (1994) Chemoautotrophic production and respiration in the hyporheic zone of a Sonoran Desert stream. In: Stanford, J.A. and Valett, H.M. (Eds.), *Proc. Second Internatl. Conf. Groundwater Ecology*, Amer. Water Res. Assoc., Bethesda, MD, pp. 329-338.

Jordan, T.E., Correll, D.L., Peterjohn, W.T. and Weller, D.E. (1986) Nutrient flux in a landscape: the Rhode River watershed and receiving waters. In: Correll, D.L. (Ed.), *Watershed Research Perspectives*, Smithsonian Press, Wash. DC, pp. 57-76.

Jordan, T.E., Correll, D.L. and Weller, D.E. (1993). Nutrient interception by a riparian forest receiving inputs from adjacent cropland. *J. Environ. Qual.*, **22**, 467-473.

Karr, J.R. and Schlosser, I.J. (1978) Water resources and the land-water interface. *Science*, **201**, 229-234.

King, J.M., Day, J.A. and Davies, B.R. (1987). Particulate organic matter in a mountain stream in the south-western Cape, South Africa. *Hydrobiologia*, **154**, 165-187.

Kitchens Jr., W.M., Dean, J.M., Stevenson, L.H. and Cooper, J.H. (1975) The Santee Swamp as a nutrient sink. In: Howell, F.G., Gentry, J.B. and Smith, M.H. (Eds.), *Mineral Cycling in Southeastern Ecosystems*, Savannah River Ecology Lab., Aiken, GA, pp. 349-366.

Klarer, D.M. and Millie, D.F. (1989) Amelioration of storm-water quality by a freshwater estuary. *Arch. Hydrobiol.*, **116**, 375-389.

Kleiss, B.A., Morris, E.E., Nix, J.F. and Barko, J.W. (1989) Modification of riverine water quality by an adjacent bottomland hardwood wetland. In: Fisk, D.W. (Ed.), *Wetlands: Concerns and Successes*, Amer. Water Resources Assoc., Bethesda, MD, USA, pp. 429-438.

Knauer, N. and Mander, U. (1989) Studies on the filtration effect of differently vegetated buffer strips along inland waters in Schleswig-Holstein. 1. Information: Filtration or nitrogen and phosphorus. *Zeit. fur Kulturtechnik und Landentwicklung*, **31**, 52-57.

LaBough, J.W. (1986) Wetland ecosystem studies from a hydrologic perspective. *Water Resources Bull.*, **22**, 1-10.

Lockaby, B.G., McNabb, K.L. and Hairston, J.E. (1994) Changes in groundwater nitrate levels across a land-use drainage continuum. In: *Riparian Ecosystems in the Humid U.S., Functions, Values and Management*, Natl. Assoc. Conserv. Districts, Wash. DC, pp. 412-421.

Lowrance, R.R., Todd, R.L. and Asmussen, L.E. (1983) Waterborne nutrient budgets for the riparian zone of an agricultural watershed. *Agricult. Ecosyst. Environ.*, **10**, 371-384.

Lowrance, R.R., Todd, R.L. and Asmussen, L.E. (1984a) Nutrient cycling in an agricultural watershed. I. Phreatic movement. *J. Environ. Qual.*, **13**, 22-27.

Lowrance, R.R., Todd, R.L. and Asmussen, L.E. (1984b) Nutrient cycling in an agricultural watershed. II. Stream flow and artificial drainage. *J. Environ. Qual.*, **13**, 27-32.

Lowrance, R.R., Todd, R.L., Fail Jr., J., Hendrickson Jr., O., Leonard, R. and Asmussen, L. (1984c) Riparian forests as nutrient filters in agricultural watersheds. *Bioscience*, **34**, 374-377.

Lowrance, R., Leonard, R. and Sheridan, J. (1985) Managing riparian ecosystems to control nonpoint pollution. *J. Soil & Water Conserv.*, **40**, 87-97.

Lowrance, R., McIntyre, S. and Lance, C. (1988) Erosion and deposition in a field/forest system estimated using Cesium-137 activity. *J. Soil & Water Conserv.*, **43**, 195-199.

Lowrance, R., Vellidis, G. and Hubbard, R.K. (1995) Denitrification in a restored riparian forest wetland. *J. Environ. Qual.*, **24**, 808-815.

Lynch, J.A. and Corbett, E.S. (1990) Evaluation of best management practices for controlling nonpoint pollution from silvicultural operations. *Water Resour. Bull.*, **26**, 41-52.

Mander, U.E., Metsur, M.O. and Kulvik, M.E. (1989) Storungen des Stoffkreislaufs, des Energieflusses und des Bios als Kriterien fur die Bestimmung der Belastung der Landschaft. *Petermanns Geographische Mitteilungen*, **133**, 233-244.

Mariotti, A., Landreau, A. and Simon, B. (1988) 15N isotope biogeochemistry and natural denitrification process in groundwater: Application to the chalk aquifer of northern France. *Geochim. Cosmochim. Acta*, **52**, 1869-1878.

McDowell, W.H. and Fisher, S.G. (1976) Autumnal processing of dissolved organic matter in a small woodland stream ecosystem. *Ecology,* **57**, 561-567.

Minshall, G.W. (1978) Autotrophy in stream ecosystems. *Bioscience,* **28**, 767-771.

Mitsch, W.J., Dorge, C.L. and Weimhoff, J.R. (1977) *Forested Wetlands for Water Resource Management in Southern Illinois.* Research Report Number 132, Univ. Illinois, Water Resources Center, Urbana, IL, 275 pp.

Mitsch, W.J., Dorge, C.L. and Wiemhoff, J.R. (1979) Ecosystem dynamics and a phosphorus budget of an alluvial cypress swamp in southern Illinois. *Ecology,* **60**, 1116-1124.

Nelson, W.M., Gold, A.J. and Groffman, P.M. (1995) Spatial and temporal variation in groundwater nitrate removal in a riparian forest. *J. Environ. Qual.,* **24**, 691-699.

Osborne, L.L. and Kovacic, D.A. (1993) Riparian vegetated buffer strips in water quality restoration and stream management. *Freshwater Biol.,* **29**, 243-258.

Parsons, J.E., Daniels, R.B., Gilliam, J.W. and Dillaha, T.A. (1994) *Reduction in sediment and chemical load in agricultural field runoff by vegetative filter strips.* Report No. UNC-WWRI-94-286, Water Resources Res. Inst., Raleigh, NC, USA, 45 pp.

Peterjohn, W.T. and Correll, D.L. (1984) Nutrient dynamics in an agricultural watershed:observations on the role of a riparian forest. *Ecology,* **65**, 1466-1475.

Peterjohn, W.T. and Correll, D.L. (1986) The effect of riparian forest on the volume and chemical composition of baseflow in an agricultural watershed. In: Correll, D.L. (Ed.), *Watershed Research Perspectives,* Smithsonian Press, Washington, DC, pp. 244-262.

Pinay, G. and Décamps, H. (1988) The role of riparian woods in regulating nitrogen fluxes between the alluvial aquifer and surface water: A conceptual model. *Regulated Rivers: Res. Manage.,* **2**, 507-516.

Pinay, G., Décamps, H., Arles, C. and Lacassin-Seres, M. (1989) Topographic influences on carbon and nitrogen dynamics in riverine woods. *Arch. Hydrobiol.,* **114**, 401-414.

Pinay, G., Roques, L. and Fabre, A. (1993) Spatial and temporal patterns of denitrification in a riparian forest. *J. Appl. Ecol.,* **30**, 581-591.

Rhode, W.A., Asmussen, L.E., Hauser, E.W., Wauchope, R.D. and Allison, H.D. (1980) Trifluralin movement in runoff from a small agricultural watershed. *J. Environ. Qual.,* **9**, 37-42.

Schultz, R.C., Colletti, J.P., Simpkins, W.W., Mize, C.W. and Thompson, M.L. (1994) Developing a multispecies riparian buffer strip agroforestry system. In: *Riparian Ecosystems in the Humid U.S., Functions, Values and Management,* Natl. Assoc. Conserv. Districts, Wash. DC, pp. 203-225.

Sedell, J.R., Triska, F.J., Hall, J.D., Anderson, N.H. and Lyford, J.H. (1974) Sources and fates of organic inputs in coniferous forest streams. In: Waring, R.H. and Edmonds, R.L. (Eds.), *Integrated Research in the Coniferous Forest Biome.* Coniferous Forest Biome, Ecosyst. Anal. Stud., US/IBP, Bull. No. 5, pp. 57-69.

Schipper, L.A., Cooper, A.B., Harfoot, C.G. and Dyck, W.J. (1994a) An inverse relationship between nitrate and ammonium in an organic riparian soil. *Soil Biol. Biochem.,* **26**, 799-800.

Schipper, L.A., Harfoot, C.G., McFarlane, P.N. and Cooper, A.B. (1994b) Anaerobic decomposition and denitrification during plant decomposition in an organic soil. *J. Environ. Qual.,* **23**, 923-928.

Schlosser, I.J. and Karr, J.R. (1981a) Riparian vegetation and channel morphology impact on spatial patterns of water quality in agricultural watersheds. *Environ. Manage.,* **5**, 233-243.

Schlosser, I.J. and Karr, J.R. (1981b) Water quality in agricultural watersheds: impact of riparian vegetation during baseflow. *Water Resources Bull.,* **17**, 233-240.

Schnabel, R.R. (1986) Nitrate concentrations in a small stream as affected by chemical and hydrologic interactions in the riparian zone. In: Correll, D.L. (Ed.), *Watershed Research Perspectives,* Smithsonian Press, Wash. DC, pp. 263-282.

Schnabel, R.R. and Stout, W.L. (1994) Denitrification loss from two Pennsylvania floodplain soils. *J. Environ. Qual.,* **23**, 344-348.

Sidle, R.C. (1986) Seasonal patterns of allochthonous debris in three riparian zones of a coastal Alaska drainage. In: Correll, D.L. (Ed.), *Watershed Research Perspectives,* Smithsonian Press, Washington, DC, pp. 283-304.

Simmons, R.C., Gold, A.J. and Groffman, P.M. (1992) Nitrate dynamics in riparian forests: Groundwater studies. *J. Environ. Qual.,* **21**, 659-665.

Sinokrot, B.A. and Stefan, H.G. (1993) H.G.. Stream temperature dynamics: measurements and modeling. *Water Resources Res.,* **29**, 2299-2312.

Staver, K.W. and Brinsfield, R.B. (1991) Monitoring agrochemical transport into shallow unconfined aquifers. In: Nash, R.G. and Leslie, A.R. (Eds.), *Groundwater Residue Sampling Design,* Amer. Chem. Soc. Symp. Series 465, Wash. DC, pp. 264-278.

Sweeney, B.W. (1993) Effects of streamside vegetation on macroinvertebrate communities of White Clay Creek in Eastern North America. *Proc. Acad. Natural Sci. Phil.,* **144**, 291-340.

Theuer, F.D., Voos, K.A. and Miller, W.J. (1984) *Instream water temperature model.* USDA Fish and Wildlife Service, Cooperative Instream Flow Service Group, Instream Flow Information Paper No. 16, Fort Collins, CO, 221 pp.

Triska, F.J., Sedell, J.R. and Gregory, S.V. (1982) Coniferous forest streams. In: Edmonds, R.L. (Ed.), *Analysis of Coniferous Forest Ecosystems in the Western United States,* US/IBP Synthesis Series 14, Dowden, Hutchinson & Ross, Stroudsburg, PA, USA, pp. 292-332.

Triska, F.J., Sedell, J.R., Cromack Jr., K., Gregory, S.V. and McCorison, F.M. (1984) Nitrogen budget for a small coniferous forest stream. *Ecol. Monogr.,* **54**, 1-344.
Triska, F.J., Kennedy, V.C., Avanzino, R.J., Zellweger, G.W. and Bencala, K.E. (1989) Retention and transport of nutrients in a third-order stream in northwestern California: hyporheic processes. *Ecology,* **70**, 1893-1905.
Triska, F.J., Kennedy, V.C., Avanzino, R.J., Zellweger, G.W. and Bencala, K.E. (1990a) In situ retention-transport response to nitrate loading and storm discharge in a third-order stream. *J. N. Am. Benthol. Soc.,* **9**, 229-239.
Triska, F.J., Duff, J.H. and Avanzino, R.J. (1990b) Influence of exchange flow between the channel and hyporheic zone on nitrate production in a small mountain stream. *Can. J. Fish. Aquat. Sci.,* **47**, 2099-2111.
Triska, F.J., Duff, J.H. and Avanzino, R.J. (1993) Patterns of hydrological exchange and nutrient transformation in the hyporheic zone of a gravel-bottom stream: Examining terrestrial-aquatic linkages. *Freshwater Biol.,* **29**, 259-274.
Vitousek, P.M. (1981) Clear-cutting and the nitrogen cycle. *Ecol. Bull. (Stockholm),* **33**, 631-642.
Warwick, J. and Hill, A.R. (1988) Nitrate depletion in the riparian zone of a small woodland stream. *Hydrobiologia,* **157**, 231-240.
Webster, J.R. (1977) Large particulate organic matter processing in stream ecosystems. In: Correll, D.L. (Ed.), *Watershed Research in Eastern North America,* Smithsonian Press, Washington, DC, pp. 505-526.
Webster, J.R., Golladay, S.W., Benfield, E.F., Meyer, J.L., Swank, W.T. and Wallace, J.B. (1992) Catchment disturbance and stream response: An overview of stream research at Coweeta Hydrologic Laboratory. In: Boon, P.J., Calow, P. and , G.E. (Eds.), *River Conservation and Management,* Wiley, Chichester, pp. 231-253.
Weller, D.E., Correll, D.L. and Jordan, T.E. (1994) Denitrification in riparian forests receiving agricultural discharges. In: Mitsch, W.J. (Ed.), *Global Wetlands: Old World and New, E*lsevier, New York, pp. 117-131.
Winterbourn, M.J. (1976) Fluxes of litter falling into a small beech forest stream. *N. Z. J. Mar. Freshwater Res*., **10**, 399-416.

The hydrological role of floodplains within the drainage basin system

3

T. P. Burt

Department of Geography, University of Durham, Science Laboratories, South Road, Durham DH1 3LE, UK.

Abstract

Floodplains play several important roles within the drainage basin system. In small to medium-sized basins (<10 000 km^2), floodplains are usually quite narrow (<1 km^2) and may act either as a conduit or a barrier to water movement – and associated sediment and solute transport – from hillslopes to the river channel. Indeed, it is generally acknowledged that, for low-order streams especially, the condition of the river and the condition of the riparian zone are intimately linked. However, the ability of a floodplain to act as a pollution buffer between farmland and the river depends fundamentally on the hydrological properties of the floodplain sediments. Permeable alluvium favours subsurface flow through the floodplain sediments, providing opportunity for processes such as denitification and deposition of suspended sediment load from surface runoff. Impermeable alluvium tends to deflect influent groundwater through aquifers below the alluvium or across the floodplain surface; in either case, the buffering capacity of the floodplain is greatly restricted. In large basins, floodplains may be several kilometres wide and hillslope inputs become less important compared to overbank flooding and storage of flood waters generated in the catchment headwaters. This paper will consider the topographical and sedimentological controls of floodplain hydrology, using results from modelling studies and field experiments in southern England to illustrate general themes.

CONTEXT

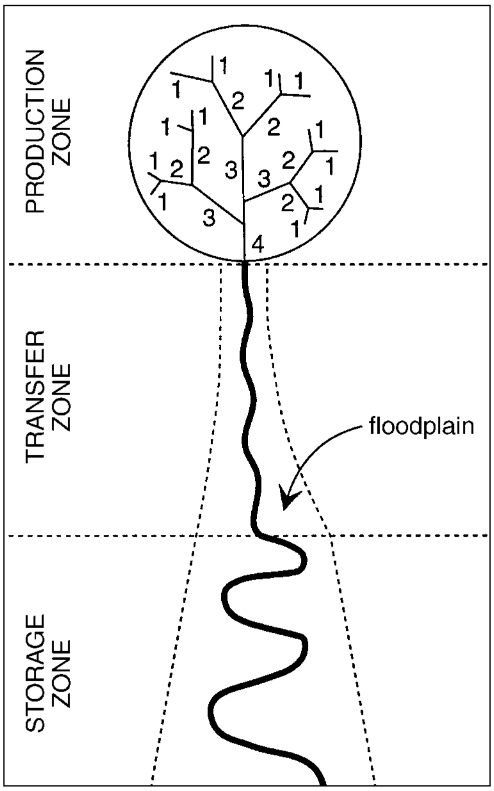
A river is usually thought of as a linear system, changing in a more or less predictable way from source to mouth. This continuum is, nevertheless, often divided for convenience into three primary zones (Schumm, 1977; Fig. 1). Headwater streams, first to third order links within the channel network (Strahler, 1952), may be viewed as the source region; here, river flow, sediment and solute load are closely coupled to hillslope processes. The middle or transfer zone, channels of fourth to sixth order, represents a transition zone; the river becomes increasingly isolated from adjacent hillslopes as floodplain width increases and the riparian land gradually becomes a sink rather than a source. The lowland floodplain river is a depositional or storage zone in which river-to-floodplain transfers dominate.

Schumm's scheme fits in well with the river continuum concept (Vannote *et al.,* 1980) which describes the progressive downstream change in stream conditions and biota, from the small, steep, turbulent and highly oxygenated headwater channels to the large, low-gradient lowland river with its diverse habitats and reliance on inputs of fine particulate organic matter from upstream and floodplain sources. However, as Petts (1994) notes, such downstream variations are more than just a structural sequence: they represent a longitudinal continuum dominated by downstream transfers of energy and matter. Moreover, it is important to emphasise the storage as well as the transfer taking place within each zone. Newbold (1994) describes how in-stream nutrient cycling occurs on a template of continual downstream transport; this open, or longitudinally displaced, cycling has been termed spiralling (Webster, 1975). Indeed, the channel network can be divided into a series of reaches or sectors, each receiving, storing and discharging water, sediments, organic matter and nutrients (Petts, 1994). It follows that biotic processes in headwater reaches (and, of course, abiotic processes too) will influence

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0 © 1997 Quest Environmental.

those in downstream reaches. In the headwaters, the supply of new nutrients from outside the channel system is very important to the aquatic ecosystem; further downstream, the products of instream cycling become dominant. A given nutrient atom, as it passes downstream may be used again and again, depending on the tightness of the spirals and the downstream displacement from one cycle to the next. This in turn depends not only on the rate of nutrient cycling itself, but also on the retentiveness of the ecosystem and the degree to which the downstream transport of a nutrient is retarded relative to that of water (Newbold, 1994). In all this, of course, the role of water as the transporting medium is fundamental.

Figure 1. Land-water interactions within a fluvial system: the three primary zones (After Schumm, 1977; Petts, 1994). Note that narrow floodplains border the channel in the production or headwater zone (c.f. Fig. 2). Further down the basin the floodplain becomes wider and hillslope inputs assume less importance relative to runoff production from the floodplain itself. In the lower reaches of the basin, the floodplain receives little water from surrounding land and inputs of flood water from the channel become of major significance to floodplain hydrology.



Newson (1994) propounds an integrated, holistic approach to river basin management, echoing earlier geomorphologists who also recognised the obvious unitary features both of geometry and process exhibited by the drainage basin (Leopold *et al.*, 1964; Chorley, 1969). Newson reminds us that the basin sediment system operates over a wide variety of timescales, from individual flood events to millennia, and re-emphasises the importance of storage as well as throughput. The sediment system has a long memory, the product of its huge storage; much of the sediment output from headwater hillslopes is stored ‘temporarily’ in channel and floodplain deposits of long residence time (Trimble, 1983). This ‘jerky conveyor belt’ of sediment delivery (Ferguson, 1981) can significantly complicate planning decisions concerning basin management. Residence times for stream solutes may often be much shorter than those for sediments, but here too, storage as well as transfer must be borne in mind, as the earlier discussion of nutrient spiralling reminds us. Nutrient transformation, for example in a buffer zone, can sometimes lead to long-term (if not permanent) retention.

The purpose of this chapter is to review the input, transfer and output of water through the drainage basin, paying particular attention to the role of floodplain land. An integrated approach clearly requires assessment of two separate but related issues. In the headwaters, runoff production is dominated by hillslope response; the narrow floodplains found there provide both a conduit and a barrier to hillslopes flows en route to the stream channel. In the middle reaches of the basin, the floodplain becomes an important source of runoff in its own right, as well as continuing to provide a conduit for upslope flows. In the lower reaches, the floodplain assumes more significance as a site for floodwater storage, especially when overbank inundation has occurred, and its role in runoff production and delivery becomes relatively less important.

It is important to stress from the outset that the essentially one-dimensional view of the drainage basin implicit in Schumm's zonation and the river continuum concept can be misleading. As Fig. 1 suggests, the river network extends throughout the basin and many first-order branches ultimately drain into a single high-order trunk. Indeed most of the channel length (and by implication, most of the basin area) is to be found in the headwaters (Strahler, 1952). This is a further reason why analysis of river basin hydrology must contrast the source region, where hillslope inputs dominate, with the downstream zone, where channel and floodplain storage are more significant.

RUNOFF PATHWAYS: FROM HILLSLOPE TO CHANNEL

Most research on hillslope hydrology has been carried out in headwater basins where slopes directly adjoin the channel, with no intervening floodplain. Further downstream, a floodplain is commonly found (Fig. 1), though curiously there has been little field research in such locations. The important point is that the hydrology of many floodplains is closely controlled by hillslope inputs; indeed if floodplains are to function as effective buffer zones, the interaction of the floodplain with slope drainage water is of crucial concern. With this in mind, a brief review of hillslope hydrology will be provided before discussing the hydrology of the floodplain itself.

Hillslope hydrology has been the subject of much research over the last three decades; full reviews are contained in Kirkby (1978) and Anderson and Burt (1990). There are many different pathways by which hillslope runoff can reach a river; these are summarised in Fig. 2. It is usual to distinguish between water which reaches the stream channel quickly, *quickflow* or *stormflow,* causing floods, and that which moves more slowly, *baseflow,* maintaining flow in rain-free periods. In practice, however, the dividing line between stormflow and baseflow during flood recession is a purely arbitrary one since subsurface drainage of soil and bedrock dominates the latter stages of flood runoff generation as well as providing baseflow (Burt, 1996).

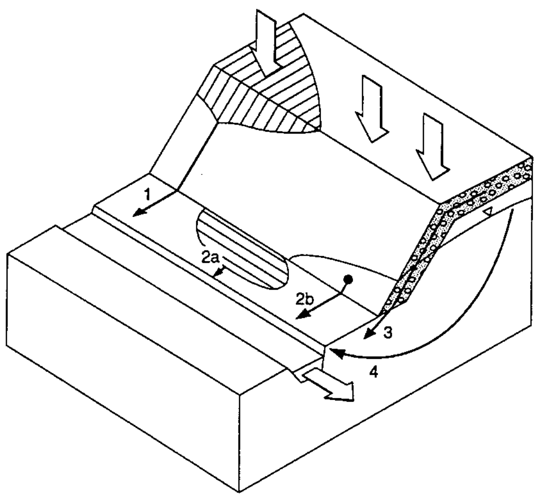
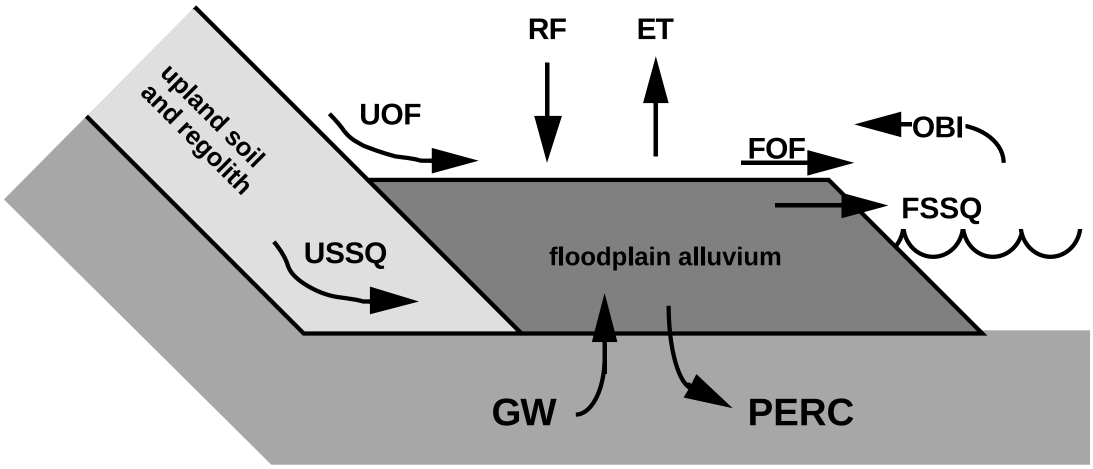


Figure 2. Hydrological pathways.
1, Infiltration-excess overland flow.
2, Saturation-excess overland flow;
2a, direct runoff; 2b, return flow.
3, Subsurface stormflow.
4, Groundwater flow. See text for discussion of these various hydrological processes.

The route by which hillslope runoff reaches the stream channel is determined by a number of factors: soil, bedrock, vegetation cover and climate (Anderson and Burt, 1990). Hydrological pathways are particularly important with respect to the speed, volume and peak rate of runoff (Fig. 3) and, perhaps more crucially, in terms of the sediment and solute load carried by the water. *Infiltration-excess overland flow* occurs when rainfall intensity exceeds the infiltration capacity of the soil; once surface depression become full, excess water overflows downslope. This stormflow mechanism has become closely associated with arable land in recent years; indeed wash and rill erosion are predominantly affected by this flow mechanism. Bare soils or those with a low crop cover are most at risk; such soils may become easily crusted during heavy rainfall, greatly increasing the likelihood of runoff and erosion (Burt and Slattery, 1996). Depending on crop type, the ‘window of opportunity’ for erosion (Boardman, 1992) may open in any season. In the UK for example, erosion of autumn-sown wheat is common in the early winter, but an increasing problem is erosion of spring-sown maize by early summer thunderstorms (Boardman *et al.*, 1995).

Figure 3. Schematic diagram of the water balance of a floodplain. Abbreviations are defined in the following section of the text.



Overland flow may also be generated when the entire soil profile becomes saturated. Here, rainfall intensity may be well below the soil’s infiltration capacity: the soil becomes saturated either because of prolonged rainfall or because of inflow from further upslope; often both processes operate together. *Return flow* occurs where water exfiltrates the soil because upslope drainage exceeds the soil’s storage capacity. Where rain falls onto a saturated soil, no infiltration can take place and direct runoff happens. Together, return flow and *direct runoff* comprise *saturation-excess overland flow.* Source areas for saturation-excess overland flow are those parts of the drainage basin where soil moisture tends to accumulate: at the bottom of any hillslope, especially where soil water converges within a hillslope hollow or where the profile shape is concave, or where the soil profile becomes thinner limiting moisture storage. The erosion of ephemeral gullies along the floors of dry (zero-order) valleys may well be encouraged by generation of saturation-excess overland flow in the valley bottom; such gullies are often the major source of sediment leaving the catchment hillslopes (Slattery *et al.*, 1994).

Subsurface flow (sometimes termed interflow or throughflow) may also drain rapidly enough from slopes to contribute to stormflow. *Subsurface stormflow* will be produced in large quantities where permeable soils overlie impermeable bedrock and where steep slopes adjoin the stream channel. Until recently, hillslope hydrologists have emphasised the importance of flow through the micropores of the soil matrix, but rapid infiltration and drainage through macropores is now recognised as a significant process in many locations too, both in relation to flood production and pollutant transport. The subsurface stormflow response tends to be more attenuated than that of overland flow; though its peak runoff rate may be lower, volumetrically it tends to dominate the flood hydrograph, particularly in the later stages, and may indeed provide a second flood peak a day or two after the main hydrograph (Anderson and Burt, 1978). Subsurface flow usually comprises long-residence soil water even where macropore flow is an important contributor, so that it invariably has a high total dissolved solids concentration (McDonnell, 1990). Continued drainage of soil and bedrock sustains streamflow during dry periods.

HYDROLOGICAL CONDITIONS WITHIN THE FLOODPLAIN

Figure 2 shows a situation in which a slope drains directly into the river channel; indeed, this is the usual configuration found in the headwater zone (Fig. 1). Further downstream, a floodplain is likely to exist, separating slope and channel. Where floodplain width remains small relative to upslope length, its main hydrological function is to provide a conduit for slope drainage. If land drainage (open ditches or tile drains) connects the slope to the channel, then the existence of the floodplain is of little consequence and slope drainage effectively short-circuits the floodplain. However, particularly in undrained state, its low gradient and waterlogged state inevitably means that the floodplain will interact with the slope drainage to some extent. As floodplain width increases, it also becomes an important source of runoff in its own right (see later section). However, before these various influences exerted by the floodplain can be discussed, it is necessary to give some details about hydrological conditions on the floodplain itself. This is, of course, a somewhat circular argument in that floodplain hydrology is not independent of slope and river, but is controlled by both to a greater or lesser extent, as the following discussion of floodplain water balance emphasises.

Given their location and structure, floodplains are likely to form wetlands, even where composed of permeable sediments (Burt, 1995). High water levels are maintained by a variety of processes: upslope discharge, upwelling groundwater, and inflows from the river channel either via bank seepage or by overbank inundation. Very low gradients across the floodplain help to sustain waterlogged conditions, especially where the floodplain is wide. Often, the alluvial deposits are fine-grained, a factor which further precludes rapid drainage. In many places, the waterlogged conditions have led to the formation of peat in the valley bottom, adding to its poorly drained state.

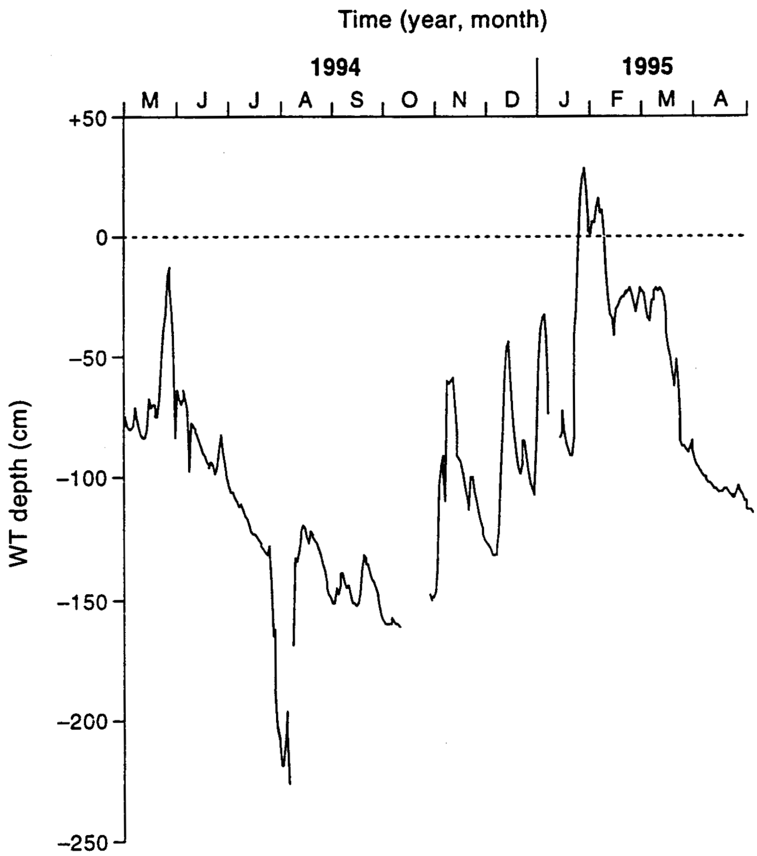
A knowledge of wetland hydrology and quantification of water inputs and outputs are necessary prerequisites to understanding wetland environments and determining their vulnerability to external influences, whether direct human impact or the indirect results of climate change (Gilvear *et al.*, in press). Nevertheless, few water balances have been computed for wetlands, those of Siegal and Glasser (1987) and Roulet (1990) being notable exceptions. Only Wassen *et al.* (1990) and Gilvear *et al.* (in press) have given water balances for floodplains. Assuming the floodplain to comprise a distinct sedimentary unit, its water balance can be defined as follows, with inputs on the left-hand side (see also figure 3):

$$\mathrm{UOF+USSQ+RF+GW+BS+OBI = FOF+FSSQ+ET+PERC+\Delta S}$$

Inputs are as follows: UOF is overland flow from the upland slope, USSQ is subsurface flow from the upland slope, RF is precipitation directly onto the floodplain, GW is groundwater discharge from the bedrock below, BS is seepage from the river channel through the bank and OBI is overbank inundation. Outputs are as follows: FOF is overland flow from the floodplain to the river, FSSQ is subsurface drainage to the river, ET is evaporation loss, and PERC is percolation to the bedrock below. Any imbalance between inputs and outputs must, by definition, involve a change in water storage within the floodplain (ΔS).

High water tables characterise most floodplains and even above the water table, the soil is likely to remain close to saturation given the upward extension of the capillary fringe (Abdul and Gillham, 1984; Gold, this volume). Figure 4 shows a typical annual cycle for water levels on the floodplain of the River Thame near Oxford, England. In summer, the water table falls significantly; this is often because of high evaporative demand rather than because of reduced inflows, especially where the floodplain sediments are impermeable and the floodplain is wide. In winter, the water balance is reversed with minimal evaporation and increased inflows and the water table rises close to ground surface. In downvalley situations (Fig. 1), the river is more likely to burst its banks during periods of high rainfall and the floodplain can remain inundated for weeks at a time.

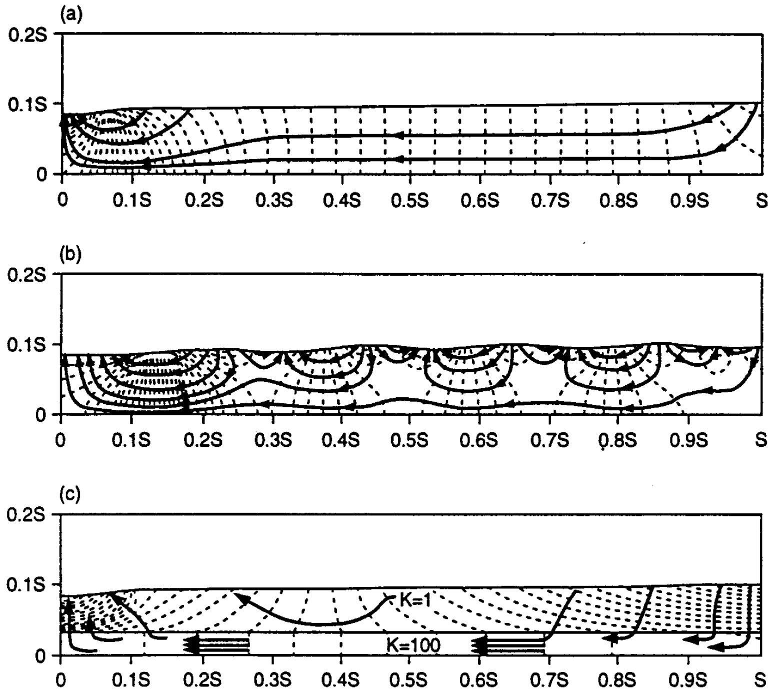
Figure 4. Annual cycle of water table fluctuation on the River Thame floodplain near Oxford, UK. A negative water table depth means that the water table lies below the ground surface; where water table depth is positive, this shows that flood water from the channel has inundated the floodplain.



FLOODPLAINS AS CONDUITS

Freeze and Witherspoon (1967) studied regional flow systems using numerical simulation for a wide upland plateau draining to a valley. This is a very similar topographic configuration to that of a floodplain draining to a river channel and for that reason their results provide a good basis for considering subsurface flow within floodplain sediments. Figure 5a shows the flow net for a simple system of homogenous permeability. The water table configuration, which closely follows the smooth topography, has a gentle slope. The hinge line (which divides the recharge from the discharge area) lies on the edge of the channel and the entire floodplain is a recharge area. Water-table slope and hydraulic gradient are greatest near the channel in the discharge zone. Throughout the rest of the floodplain, the flow pattern is relatively uniform. Figure 5b shows the water table and flow pattern which results in hummocky terrain, as might arise where the floodplain comprises infilled oxbows. There are numerous sub-basins and water may be discharged locally to the nearest topographic low or flow towards the main channel. The effect of a layer of high hydraulic conductivity is shown in figure 5c. The permeable layer forms a conduit in which flow is concentrated with recharge from the overlying alluvium and a steep upward discharge to the channel.

Figure 5. Flow nets for a floodplain-channel system (After Freeze and Witherspoon, 1967): (a) for a homogenous sedimentary layer and flat terrain; (b) for a homogenous sedimentary layer and hummocky terrain; (c) for a floodplain with a more permeable layer at depth showing bypass flow beneath the upper layer.



In many cases, the floodplain is coupled to its hillslope and the combined drainage system must be considered. Figure 6 summarises the ways in which water can flow across a floodplain to reach the stream channel. Overland flow moving on to the floodplain will infiltrate provided that sufficient storage capacity exists i.e. the water table is not at or close to the ground surface. Infiltration will be maximised where the infiltration capacity of the floodplain soil is high, and *vice versa.* If the floodplain is completely waterlogged, overland flow will move rapidly across the floodplain to the river (Waddington *et al.,* 1993). Subsurface slope drainage may follow one of three main pathways:

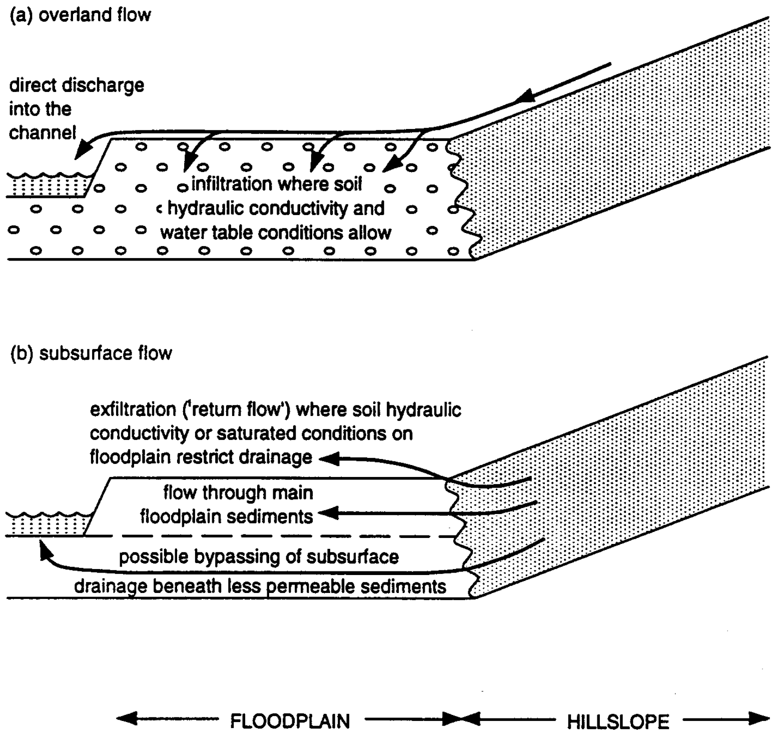
1. Where the floodplain sediments are impermeable, but there is a permeable layer at depth, either a gravel deposit below the alluvium or a permeable bedrock, subsurface flow can move quickly under the floodplain alluvium to the stream (cf. Fig. 5c). This is thought to be a very common situation in British floodplains where late Pleistocene gravels are overlain by less permeable Holocene alluvium. There is relatively little flow through the upper alluvium which, though remaining largely saturated, remains somewhat stagnant. In such cases, the potential of the floodplain to act as a nitrate buffer zone may be greatly reduced. A similar example is given in Plenet and Gibert (1992). Examples of deep groundwater upwelling to recharge a valley-bottom wetland are provided by Lloyd and Tellam (1995) and Gilvear *et al.* (in press).

2. In the absence of a permeable substrate, slope drainage is forced to the surface, moving as overland flow to the channel. Waddington *et al.* (1993) studied a riparian wetland near Toronto, Canada. Springs emerging at the wetland margin produce rivulets which flow across the riparian zone to the spring. This is a major discharge pathway with the result that much of the wetland's subsurface buffer potential is effectively bypassed. Where the floodplain terrain consists of hummocks and pools (cf. Fig. 5b), the pattern of surface flow across the floodplain may be complex (cf. Burt and Gardiner, 1984).

3. Only where floodplain sediments are permeable and homogenous will slope water move through the alluvium to the stream in a uniform, unconcentrated manner (cf. Fig. 5a). In terms of buffer zone

functions which require interaction between water and soil, only this last situation will provide the optimal hydrological condition. Correll (this volume) shows that the depth of the permeable layer is critical: if subsurface flow is too deep, the saturated zone lies well below the soil and the buffer zone is unable to function effectively. Optimally, the depth of permeable substrate above an aquiclude should be sufficient to fully saturate the soil horizons, but without generating overland flow.

Figure 6. Summary of main flow paths by which hillslope discharge moves through a floodplain to reach the channel: (a) for surface water inputs from upslope; (b) for subsurface water inputs from upslope.



Many field studies have used only a single line of piezometers to monitor flow across the floodplain (e.g. Grieve *et al.*, 1995). In most cases the hydraulic gradient does not lie orthogonal to the channel but is aligned downvalley in response to the floodplain gradient (Anderson and Kneale, 1982). In order to sample a given body of water as it moves across the floodplain (for example, to observe progressive loss of nitrate), it is necessary to adopt a three-dimensional approach using a grid of piezometers (see for example, Haycock and Burt, 1993). A single row of piezometers may involve waters of different origin and false conclusions could be made concerning the evolution of the water quality as a result. Sedimentary structures such as infilled oxbows may also complicate flow patterns on a floodplain, again requiring a three-dimensional survey of water table height. Vegetation patch mosaics found on floodplains imply that sedimentary structures and resulting flow paths may be very complex in some instances (Large and Petts, 1994). It is important to emphasise therefore that water does not flow across the floodplain in a simple and uniform manner. Flow paths may well be tortuous and will very often tend in a downvalley direction.

An important outcome is that the pattern of groundwater discharge along a stream will be spatially variable, with certain locations forming important inflow sites (Anderson and Burt, 1978). In headwater basins, diffuse hillslope runoff will tend to concentrate in hillslope hollows (or swales); these are the variable source areas discussed earlier. In such cases non-point (or diffuse) pollution may flow across the floodplain as concentrated (overland) flow and enter the river channel at a single point source. These sources must clearly form the focus of attention in any buffer zone scheme. A further cause of spatial variation in inflows relates to the thickness and permeability of the floodplain sediments. Where these are thin or impermeable, surface runoff is likely. Thicker, permeable sediments tend to favour subsurface flow and where there is an aquifer below, saturated conditions may occur only at great depth, with important consequences for buffer zone functions (see Correll, this volume).

FLOODPLAINS AS A SOURCE OF RUNOFF

Once floodplains become sufficiently wide, they become an important source of runoff in their own right as well as continuing to provide a conduit for slope drainage to the river. Undrained floodplains may well be waterlogged to the surface and may therefore be an important source of saturation-excess overland flow; under such circumstances, spring water emerging from the base of the hillslope will be conducted rapidly across the floodplain surface to the stream.

In many cases, floodplains with heavy clay soils have been drained as a means of making waterlogged soils more manageable, either by open ditches or by underdrainage (often called tile drainage in the UK). Decrease in water content, and therefore an increase in bearing strength, allows better access and more opportunities for cultivation (Parkinson, 1988). As a result, many floodplains, formerly used only for summer grazing, are now under continuous arable cultivation. Drainage has two effects: firstly, hillslope water moves rapidly to the stream with no opportunity for interaction with floodplain soils; in essence the slope becomes directly coupled to the channel. Secondly, the drained floodplain itself becomes an important source of subsurface water which, in combination with more intensive land use, may result in large losses of soluble soil nutrients from near-stream land. Green (1979) perceptively linked some of the increase in nitrate concentrations seen in English rivers since the 1960s to the extensive drainage of floodplains and clay vales. Clay soils are often macroporous; this in combination with underdrainage results in very rapid disposal of rainfall (Robinson and Beven, 1983) giving little or no opportunity for soil buffering as the water drains to the river.

RIVER-TO-FLOODPLAIN FLOWS

There are two occasions when the normal hillslope-to-floodplain flow direction may reverse:

1. As noted above, in the upper reaches of a catchment, subsurface contributions to streamflow aid in the build-up of a flood wave. In the lower reaches during a flood the river stage may be higher than the floodplain water table and river water will then flow into the floodplain sediments; this is often referred to as bank storage (Freeze and Cherry, 1979). The impact of bank storage in moderating the flood hydrograph is shown on figure 7. As the flood wave arrives, flow is induced into the bank; as the stage declines, the flow is reversed and the stored water flows back out into the channel. The effect is to attenuate the flood wave, reducing and delaying the peak discharge. A more extreme version of this effect occurs when the river bursts its banks and inundates the floodplain. This in itself causes large quantities of floodwater to be stored. Further storage takes place because some floodwater then infiltrates the floodplain soil. Current research at Bristol and Durham Universities aims to investigate these processes: the two-dimensional modelling schemes of Bates and Anderson (1993) will be extended by linking a one-dimensional finite difference model of subsurface hydrology to the two-dimensional finite element model of floodplain/channel flows. In addition, water fluxes to and from the hillslope will be simulated by integrating this scheme with a three-dimensional finite difference hillslope hydrology model. Hitherto, all current hydraulic models have had closed impermeable boundaries except at the inflow and outflow points to the modelled reach; in effect no flux of water occurs between the channel/floodplain flow and floodplain soils or adjacent hillslopes. As well as its impact on the flood hydrograph, overbank inundation may provide an additional pathway by which a floodplain may act as a sink for sediment-bound contaminants and nutrients. Current work by Price (in prep.) aims to couple a denitrification routine to the one-dimensional infiltration model mentioned earlier.

2. Under low-flow conditions, groundwater recharge by influent seepage from a river channel is a common occurrence. In many cases, flow from the hillslope across the floodplain to the channel takes place only in the winter months. In summer, the hydraulic gradient is reversed and water moves from channel to floodplain. Haycock (1991) has monitored this seasonal reversal of the hydraulic gradient for a small floodplain near Oxford (Fig. 8). In summer, the water table in the local limestone aquifer falls sufficiently so that inflow from the river takes place; once the aquifer has been recharged in the autumn, the aquifer then drains through the floodplain sediments to the channel. The period of hillslope-to-floodplain flow thus coincides with the time of maximum discharge and peak nitrate concentrations in subsurface flow and the floodplain is able to function as a nitrate buffer zone.

Figure 7. Flood wave modification due to bank storage effects (based on Freeze and Cherry, 1979): (a) change in flood stage during the flood; (b) attenuation of flood hydrograph because of bank storage; (c) flow into and out of the channel bank during the flood; (d) volume of flood water stored in the channel bank; (e) the impact of flow reversal on groundwater discharge to the flood hydrograph.

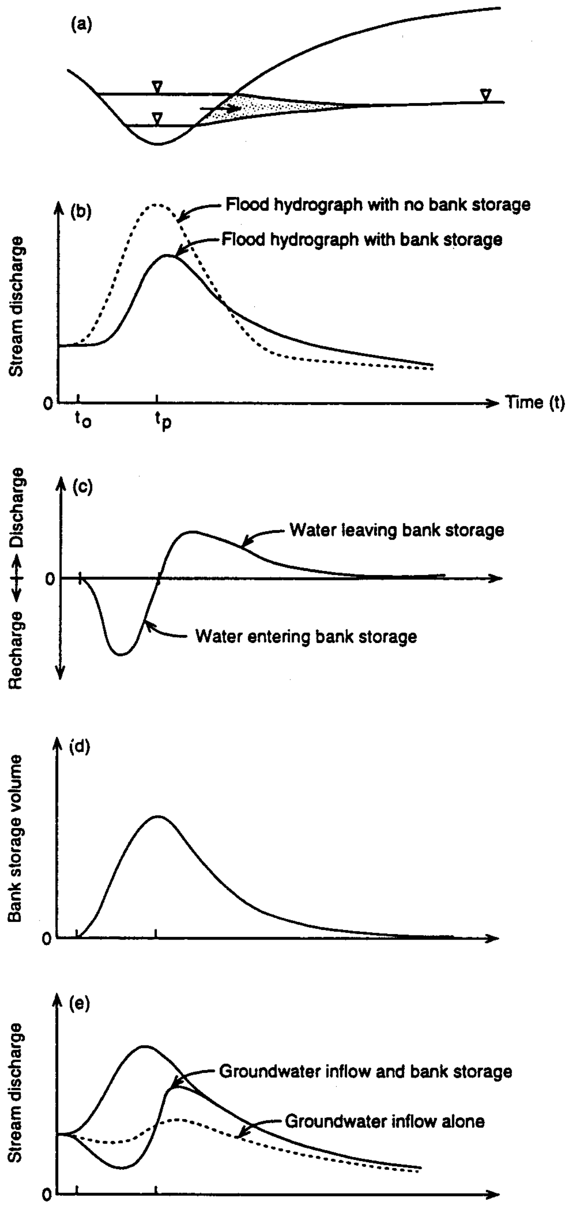
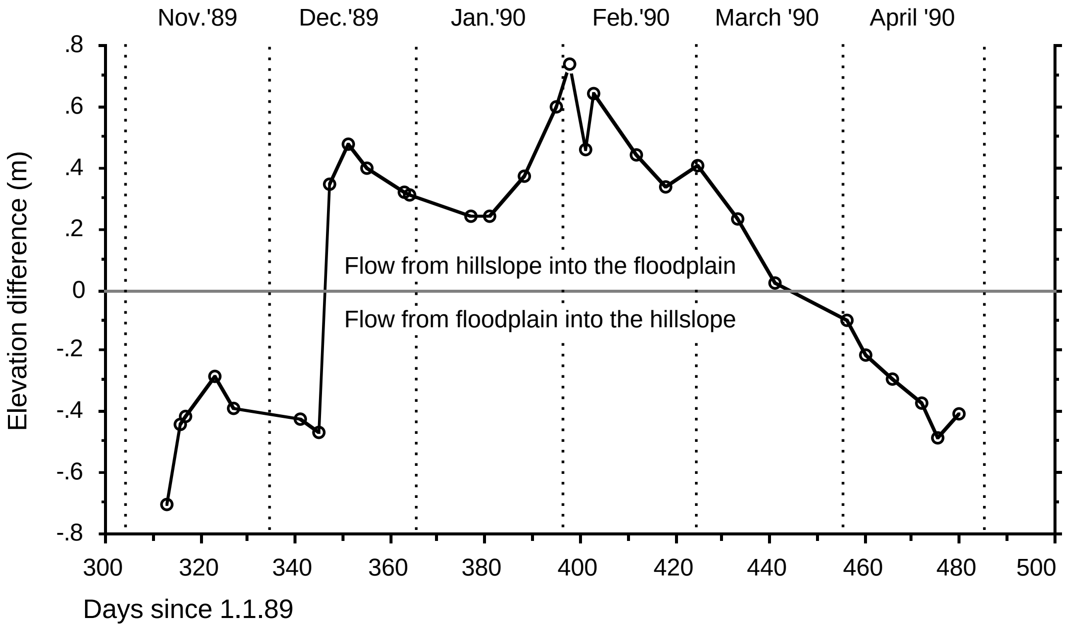


Figure 8. Seasonal reversal of the hydraulic gradient across the floodplain of the River Leach near Oxford (reproduced from Haycock, 1991, by kind permission of the author). The graph shows the difference in water table elevation potential between row 4 and row 6 (distance = 11 m) at the grass floodplain site described by Haycock (1991; see also Haycock and Burt, 1993). In summer there is effluent seepage from the channel into the floodplain; in winter, the flow direction is reversed and groundwater discharges into the channel.



In both instances, river-to-floodplain flows assume more importance in the lower reaches of the drainage basin (Fig. 1) so that the floodplain becomes more important as a sink than a source of water, sediment and nutrients. Many buffer zone processes operate most effectively in headwater basins where hillslope-to-channel flows dominate and where most of the channel length (and by implication, most of the riparian land) is located. However, though the opportunity for buffer zone functions may be less in the downstream reaches of the basin, the importance of floodplain and channel storage in that zone should not be underestimated.

REFERENCES

Abdul, A.S. and Gillham, R.W. (1984) Laboratory studies of the effects of the capillary fringe on streamflow generation. *Water Resources Research* **20**, 691-8.

Anderson, M.G. and Burt, T.P. (1978) The role of topography in controlling throughflow generation. *Earth Surface Processes,* **3**, 331-344.

Anderson, M.G. and Burt, T.P. (1990) *Process Studies in Hillslope Hydrology.* John Wiley & Sons: Chichester.

Anderson, M.G. and Kneale, P.E. (1982) The influence of low-angled topography on hillslope soil water convergence and stream discharge. *Journal of Hydrology,* **57**, 65-80.

Bates, P.D. and Anderson, M.G. (1993) A two dimensional finite element model for river flow inundation. *Proceedings of the Royal Society of London Series A,* **440**, 481-91.

Boardman, J. (1992) Agriculture and erosion in Britain. *Geography Review,* **6**, 15-19.

Boardman, J., Burt, T.P., Slattery, M.C., Evans, R. and Shuttleworth, H. (1995) Soil erosion and flooding as a result of a summer thunderstorm in Oxfordshire and Berkshire, May 1993. *Applied Geography,* **16**, 21-34.

Burt, T.P. (1995) The role of wetlands in runoff generation from headwater catchments. In: Hughes, J.M.R. and Heathwaite, A.L. (Eds.) *Hydrology and Hydrochemistry of British Wetlands,* John Wiley: Chichester, 21-38.

Burt, T.P. (1996) The hydrology of headwater catchments. In: Petts, G.E. and Calow, P. (Eds.) *River Flows and Channel Forms,* Blackwell: Oxford, 6-31.

Burt, T.P. and Gardiner, A.T. (1984) Runoff and sediment production in a small peat-covered catchment: some preliminary results. In Burt, T.P. and Walling, D.E. (Eds.) *Catchment Experiments in Fluvial Geomorphology,* Geobooks: Norwich, 133-52.

Burt, T.P. and Slattery, M.C. (1996) Time-dependent changes in soil surface condition and runoff generation. In: Anderson, M.G. and Brooks, S.M. (Eds.), *Advances in Hillslope Processes,* John Wiley: Chichester, in press.

Chorley, R.J. (1969) The drainage basin as the fundamental geomorphic unit. In: Chorley, R.J. (Ed.) *Water, Earth and Man,* Methuen: London, 37-59.

Ferguson, R.I. (1981) Channel form and changes. In: Lewin, J. (Ed.), *British Rivers,* Allen & Unwin: London, 90-125.

Freeze, R.A. and Cherry, J.A. (1979) *Groundwater.* Prentice-Hall: Englewood Cliffs.

Freeze, R.A. and Witherspoon, P.A. (1967) Theoretical analysis of regional groundwater flow: 1. Analytical and numerical solutions to the mathematical model. *Water Resources Research*, **2**, 641-656.

Gilvear, D.J., Sadler, P.J.K., Tellam, J.H. and Lloyd, J.W. (in press). Surface water processes and groundwater flow within a hydrologically complex floodplain wetland, Norfolk Broads, UK. *Journal of Hydrology,* in press.

Green, F.H.W. (1979). Land drainage. In: Hollis, G.E. (Ed.) *Man's Role in the Hydrological Cycle of the UK.* Geobooks: Norwich.

Grieve, I.C., Gilvear, D.J. and Bryant, R.G. (1995) Hydrochemical and water source variations across a floodplain mire, Insh Marshes, Scotland. *Hydrological Processes,* **9**, 99-110.

Haycock, N.E. (1991) *Riparian Land as Buffer Zones in Agricultural Catchments.* Unpublished D.Phil. thesis, University of Oxford.

Haycock, N.E. and Burt, T.P. (1993) The sensitivity of rivers to nitrate leaching: the effectiveness of near-stream land as a nutrient retention zone. In: Allison, R.J. and Thomas, D.S.G. (Eds.), *Landscape Sensitivity,* John Wiley: Chichester, 261-272.

Kirkby, M.J. (1978) *Hillslope Hydrology.* John Wiley: Chichester.

Large, A.R.G. and Petts, G.E. (1994) Rehabilitation of river margins. In: Petts, G.E. and Calow, P. (Eds.), *The Rivers Handbook Volume Two,* Blackwell: Oxford, 401-418.

Leopold, L.B., Wolman, M.G. and Miller, J.P. (1964) *Fluvial Processes in Geomorphology.* Freeman: San Francisco.

Lloyd, J.W. and Tellam, J.H. (1995) Groundwater-fed wetlands in the UK. In: Hughes, J.M.R. and Heathwaite, A.L. (Eds.), *Hydrology and Hydrochemistry of British Wetlands,* John Wiley: Chichester, 39-61.

McDonnell, J.J. (1990) A rationale for old water discharge through macropores in a steep, humid catchment. *Water Resources Research,* **26**, 2821-32.

Newbold, J.D. (1994) Cycles and spirals of nutrients. In: Petts, G.E. and Calow, P. (Eds.), *The Rivers Handbook Volume One,* Blackwell: Oxford, 379-408.

Newson, M.D. (1994) Sustainable integrated development and the basin sediment system: guidance from fluvial geomorphology. In: Kirby, C. and White, W.R. (Eds.), *Integrated River Basin Development,* John Wiley: Chichester, 1-10.

Parkinson, R.J. (1988) Field drainage and the soil environment. *Outlook on Agriculture,* **17**, 140-145.

Petts, G.E. (1994) Rivers: dynamic components of catchment ecosystems. In: Petts, G.E. and Calow, P. (Eds.), *The Rivers Handbook Volume Two,* Blackwell: Oxford, 3-22.

Plenet, S. and Gibert, J. (1992) A floodplain spring: an ecotone between surface water and groundwater. *Regulated Rivers,* **7**, 93-102.

Robinson, M. and Beven, K.J. (1983) The effect of mole drainage on the hydrological response of a swelling clay soil. *Journal of Hydrology,* **64**, 205-23.

Roulet, N. (1990) Hydrology of a headwater basin wetland: groundwater discharge and wetland maintenance. *Hydrological Processes,* **4**, 387-400.

Schumm, S.A. (1977) *The Fluvial System.* John Wiley: New York.

Siegal, D.I. and Glasser, P.H. (1987) Groundwater flow in a bog-fen complex, Lost River peatland, Northern Minnesota. *Journal of Ecology,* **75**, 743-54.

Slattery, M.C., Burt, T.P. and Boardman, J. (1994) Rill erosion along the thalweg of a hillslope hollow: a case study from the Cotswold Hills, Central England. *Earth Surface Processes and Landforms,* **19**, 377-86.

Strahler, A.N. (1952) Quantitative analysis of watershed geomorphology. *American Geophysical Union Transactions,* **38**, 913-20.

Trimble, S.W. (1983) A sediment budget for Coon Creek basin in the Driftless Area, Wisconsin, 1853-1977. *American Journal of Science,* **277**, 876-87.

Vannote, R.L., Minshall, G.W., Cummins, K.W., Sedell, J.R. and Cushing, C.E. (1980). The river continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences,* **37**, 130-7.

Waddington, J.M., Roulet, N. and Hill, A.R. (1993) Runoff mechanisms in a forested groundwater discharge wetland. *Journal of Hydrology,* **147**, 37-60.

Wassen, M.J., Barendregt, A., Palczynski, A., de Smidt, J.T. and de Mars, H. (1990) The relationship between fen vegetation gradients, groundwater flow and flooding in an undrained valley mire at Biebrza, Poland. *Journal of Ecology,* **78**, 1106-22.

Webster, J.R. (1975) *Analysis of potassium and calcium dynamics in stream ecosystems on three southern Appalachian watersheds of contrasting vegetation.* Unpublished Ph.D. thesis, University of Georgia, Athens.

Buffer Zones as Sediment Traps or Sources

4

T.A. Dillaha III[1] and S.P. Inamdar[2]

[1] *Department of Biological Systems Engineering, Virginia Polytechnic Institute and State University, Blacksburg, VA 24061-0303, USA.*
[2] *USDA-ARS Southeast Watershed Research Laboratory, P.O. Box 946, Tifton, GA 31793, USA.*

Abstract

Riparian buffer zones are generally considered to be sediment sinks in watersheds disturbed by human activities. However, under some circumstances, buffer zones may be sediment sources. This paper describes the principal processes affecting sediment transport to and within buffer zones, reported trapping efficiencies for various types of buffer zones, techniques for enhancing the effectiveness of buffer zones as sediment sinks, maintenance requirements and the likely long-term effectiveness of buffer zones for sediment trapping. Both herbaceous and forested buffer zones are considered, as well as three-zoned buffers consisting of unmanaged and managed forest zones and a managed herbaceous zone immediately adjacent to the runoff source area. Effects of changes in watershed hydrology as well as the effects of sediment accumulations and concentrated flow on buffer performance are discussed along with models that have been developed to describe sediment transport in buffer zones.

INTRODUCTION

Non-point source (NPS) pollution from agricultural and urban areas is now recognised as one of the most significant water quality problems facing the world. In 1995, the US Environmental Protection Agency (USEPA) reported that 40% of US rivers, lakes and estuaries did not meet water quality requirements (USEPA, 1995). Non-point source pollution from stormwater runoff and snowmelt was identified as the leading cause of impairment in rivers, lakes and estuaries. Sediment was identified as the primary pollutant in rivers and the second most important pollutant in lakes. Agriculture was identified as the primary source of pollution in rivers and lakes and the main source of sediment in all water bodies. Magnitudes and sources of water quality impairment in other industrialised countries, where point source pollution has been controlled, are similar.

To help reduce sediment, nutrient, pesticide and pathogen loadings to surface waters, environmentalists and government agencies have been promoting riparian buffer zones as a means of removing pollutants from surface and subsurface stormwater runoff. In addition, buffer zones promote channel stability, reduce flooding and enhance aquatic habitats. Buffer zones are bands of planted or indigenous vegetation, situated between pollutant source areas and ephemeral and perennial streams and other waterbodies. Pollutant removal in buffer zones is accomplished by a combination of physical, chemical and biological processes. These processes are poorly understood, but there is a general consensus in the scientific community that buffer zones are an essential element of all NPS pollution control programmes (Karr and Schlosser, 1978; Gilliam, 1994; Lowrance *et al.*, 1995). Unfortunately, there is considerable uncertainty as to the site-specific effectiveness of buffer zones in removing pollutants from runoff and there are no standards or accepted methods for buffer zone design. As a consequence, buffer size requirements are typically established by political acceptability, not scientific merit. Buffers that are undersized place aquatic resources at risk; while buffers that are larger than needed unnecessarily deny landowners the use of a portion of their land (Castelle *et al.*, 1994).

The primary goal of this paper is to examine the role of buffer zones in reducing sediment loads to receiving waters. Excessive erosion and sedimentation results in destruction of aquatic habitats;

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0 © 1997 Quest Environmental.

increased sediment-bound pollutant transport, water turbidity, water treatment costs and flooding and decreased recreational water use and water storage capacity. In the US, soil erosion is estimated to cause in excess of $6 billion in annual off-site damages (Clark, 1985) and over $1 billion in annual on-site damages due to declines in crop productivity (Baker, 1992). These costs were estimated in the 1980s and have almost certainly escalated since then.

This paper has three specific objectives: 1) To review the literature pertaining to the effectiveness of buffer zones in removing sediment from runoff, 2) To discuss how buffer zones can sometimes become sediment sources, and 3) To suggest ways in which the buffer zone sediment trapping can be enhanced.

REVIEW OF BUFFER ZONE EFFECTIVENESS

The use of buffer zones as a primary means of removing sediment and other pollutants from runoff is a relatively new practice. Historically, sediment control efforts minimised off-site pollution by reducing upland erosion and surface runoff. Buffer zones, on the other hand, are designed to remove sediment from runoff once it has left the upland area. The major sediment removal mechanisms associated with buffer zones involve changes in flow hydraulics which enhance the opportunity for the infiltration of runoff and fine sediment into the soil, sediment deposition and filtration of sediment by vegetation. For these mechanisms to be effective, it is essential that the surface runoff passes slowly through the buffer to provide sufficient contact time for the removal mechanisms to function.

Infiltration is one of the most significant removal mechanisms affecting buffer zone performance. Infiltration is important since the finer sediment particles enter the soil profile along with infiltrating water and because it decreases surface runoff, thus reducing sediment transport capacity. Since infiltration is one of the more easily quantifiable mechanisms affecting buffer zone performance, many buffer zones have been designed to allow all runoff from a design storm to infiltrate into the buffer zone. This approach results in large land requirements because it ignores other removal mechanisms.

Buffer zones also remove sediment through the process of deposition. Buffer zones are usually composed of either dense herbaceous vegetation or forests with sparse understorey and dense surface litter. These surface conditions offer high resistance to shallow overland flow and decrease the velocity of overland flow immediately upslope and within the buffer zone, causing significant reductions in sediment transport capacity. If the transport capacity is less than the incoming sediment load, then the excess sediment may be deposited and trapped. The filtration of sediment by vegetation during overland flow is not as well-understood as the infiltration and deposition processes. Infiltration is probably most significant for clay sized particles while deposition is most significant for silt and larger sized particles. Filtration is significant only with the largest soil particles and aggregates.

Grass buffers

Wilson (1967) conducted one of the first buffer zone sediment trapping studies. He reported optimum distances required to trap sand, silt and clay in flood waters on flat slopes and concluded that buffer length, sediment load, flow rate, slope, grass height and density, and degree of vegetative submergence all affect sediment removal. Neibling and Alberts (1979) used a rainfall simulator on grass plots with a slope of 7% to show that 0.6 to 4.9 m long grass buffers reduced sediment discharge by over 90%. Clay transport was reduced by 37, 78, 82 and 83%, for the 0.6, 1.2, 2.4 and 4.9 m buffers, respectively. Significant deposition of solids was observed just upslope of the leading edge of the buffer zone and 91% of the incoming sediment load was removed within the first 0.6 m of the buffer zone.

The most comprehensive research on sediment transport in grass buffers was conducted at the University of Kentucky (Barfield *et al.*, 1979; Hayes *et al.*, 1979; Tollner *et al.*, 1982). Design equations were developed relating the fraction of sediment trapped in vegetation to the mean flow velocity, flow depth, particle fall velocity, buffer length and the spacing hydraulic radius. High trapping efficiencies were reported as long as the vegetation was not submerged, but trapping efficiency decreased dramatically at higher runoff rates which inundated the media. The Kentucky researchers, like Neibling and Alberts (1979), observed that the majority of sediment deposition occurred just upslope of the buffer and within the first metre of the buffer, until the upper portions of the buffer were buried in sediment. Subsequent flow of sediment into the buffer resulted in the advance of a wedge-shaped deposit of sediment down through the buffer. The Kentucky researchers did not consider the long-term effectiveness of buffers.

Young *et al.* (1980) used a rainfall simulator to study the ability of 27.4 m grass buffers with 4% slopes to control pollution from feedlot runoff. Sediment losses were reduced by 66 to 82%. Magette *et al.* (1989) used a rainfall simulator on field plots to study the effectiveness of 4.6 and 9.2 m grass buffers in removing nutrients and sediment from agricultural runoff. Sediment losses were reduced 52 and 75% by the 4.6 and 9.2 m buffers, respectively. Buffer zone effectiveness was also reported to decrease with time and with decreasing buffer zone to source area ratio. Dillaha *et al.* (1989a) used a rainfall simulator to evaluate the effectiveness of grass buffers for sediment and nutrient trapping. Plots were constructed with both shallow uniform flow and concentrated or channelised flow. The 9.1 and 4.6 m buffers with shallow uniform flow removed 87 and 75% of the incoming sediment. Buffers with concentrated flow were much less effective, with percentage reductions averaging 23 to 37% less for sediment.

The effectiveness of existing grass buffer zones in Virginia was qualitatively evaluated by visiting and observing buffers on 18 farms in Virginia over a 13-month period (Dillaha *et al.*, 1989b). Buffers were evaluated by talking with landowners and conservationists and observing site conditions. All the buffers were approximately 6 m in length and were used in combination with cropland. Most were installed for water quality improvement in conjunction with Virginia's Chesapeake Bay Programme. Buffer performance was generally judged to fall into two categories depending on site topography. In hilly areas, grass buffers were judged ineffective for sediment trapping because drainage usually concentrated in natural drainageways within the fields before reaching the buffers. Flow across these buffers during larger runoff-producing storms (the most significant in terms of sediment loss) was therefore primarily concentrated and the buffers were locally inundated and ineffective. This assessment was confirmed by the fact that little sediment was observed to have accumulated in the majority of the buffers observed. Buffers in these areas, while not effective for trapping sediment, were beneficial because they provided cover in areas adjacent to streams, which are often susceptible to severe localised channel and gully erosion. They also provided a narrow buffer between cropland and streams that reduces aerial drift of fertilisers and pesticides to streams during application. The effects of buffers in trapping sediment deposition during channel overflow and floodplain inundation events was not considered.

In flatter areas, such as the Coastal Plain, buffers appeared to be more effective. Slopes were more uniform and significant portions of runoff entered the buffers as shallow uniform flow. Most sediment was observed to deposit just upslope of, or within the first metre of, the field/buffer interface due to the abrupt increase in vegetation density that slows water and reduces its sediment transport capacity. The coarser sediment deposited at the field edge often formed a berm that blocked further inflow of surface runoff into the buffer zone at this point. Several buffers were observed that had trapped so much sediment that they were higher than the adjacent fields. In these cases, runoff flowed parallel to the buffer until a low point was reached where runoff crossed the buffer as concentrated flow. In this situation, the buffer acted more like a terrace. Flow parallel to the buffer zone also was observed on several farms where mouldboard ploughing was practised. When soil was turn-ploughed away from the buffer, a shallow ditch was formed parallel to the field. If this ditch was not removed later by careful disking, runoff concentrated and flowed parallel to the buffer until it reached a low point and crossed as channel flow. Berms of this type were observed with both forest and grass buffers. In response to this problem, one conservationist required landowners participating in the state buffer strip cost-share programme to construct berms or water bars perpendicular to and at 15 to 30 m intervals along width of the buffers. The water bars minimised flow parallel to the buffer and encouraged more uniform distribution of runoff through the buffer. The water bars, which were made with a tobacco ridger, were broad and shallow enough that they did not interfere with planting and harvesting operations. Water bars of this type would be particularly useful in forest buffers to distribute runoff more evenly across the width of the buffer, rather than allowing channelised flow parallel to the buffer until a developed channel is encountered. Water bars are a common forestry conservation practice used to minimise erosion on logging roads.

Most of the grass buffers observed by Dillaha *et al.* (1989b) had significant forest buffers between them and receiving waters. In many cases, surface runoff was able to pass through the forest buffer in well-developed channels with little opportunity for sediment deposition. In other cases (lower sloped areas in the Coastal Plain), well-developed channels seemed to disappear as they moved into the forested area and high rates of sediment trapping would be expected. Conclusions drawn from the plot and field observations include (Dillaha *et al.*, 1989a; 1989b):

1. Buffers of reasonable length are effective for sediment removal only if flow is shallow and uniform and if the buffers have not been previously inundated with sediment.

2. The effectiveness of herbaceous buffers for sediment removal appears to decrease with time as sediment accumulates in the buffer and encourages concentrated flow across the buffer.

3. Active maintenance is required for sustainable sediment trapping in herbaceous buffers.

a) At sites with significant flow parallel to the buffer, water bars should be constructed perpendicular to the buffer at 15-30 m intervals to intercept runoff and force it to flow through the buffer before it can concentrate further.

b) To promote vegetative growth and sediment trapping, herbaceous buffers should be mowed and the residue harvested 2-3 times per year. Mowing and harvesting of vegetation will increase vegetation density at ground level, reduce sediment transport and remove nutrients from the system.

c) Caution should be used when applying herbicides to adjacent fields to prevent accidental damage to buffer vegetation.

d) Buffers should not be used for roadways or turn rows because traffic will damage the buffers and may cause concentrated flow problems.

e) Cattle should be excluded from buffers at all times, but especially during periods when soils are moist and buffers are most susceptible to hoof damage.

f) Buffers should be inspected regularly for damage caused by farming operations and should be repaired as soon as possible.

g) Buffers that have accumulated excessive sediment must be ploughed out, disked and graded if necessary and re-seeded, in order to re-establish shallow sheet flow conditions.

h) Extreme care must be taken during tillage operations to avoid reducing the length of the buffer. If mouldboard ploughing is practised, the last plough pass should turn soil toward the buffer and the disturbed area next to the buffer should be carefully disked to minimise gully formation and concentrated flow parallel to the buffer.

4. Most on-farm herbaceous buffers observed were judged to be ineffective for sediment removal because most flow tended to accumulate in natural drainageways before reaching the buffer zone. This was more of a problem in hilly areas and less of a problem in flatter areas such as the coastal plain.

5. Buffers should be installed on the contour as much as possible to promote shallow uniform flow across the buffer.

6. Large fields with significant natural drainageways or grassed waterways are acceptable for buffers only if buffers are installed on both sides of internal field drainageways. This will trap sediment before it can enter the drainageways.

7. If slope-lengths are not excessive and buffers are on the contour, up and down slope tillage (perpendicular to the buffer) is preferred over contour tillage because it will distribute runoff more uniformly along the length of the buffer. Unfortunately, it is difficult to till up and downslope without damaging the buffer because of the necessity of using it as a turn row.

Dillaha *et al.* (1989a) concluded that the effectiveness of experimental buffer zones with shallow uniform flow, which are typically used by researchers in short-term studies, should not be used as a direct indicator of real world buffer zone effectiveness because of long-term sediment accumulation and concentrated flow problems previously discussed. Concentrated flow effects under real agricultural conditions were estimated to be orders of magnitude greater than those encountered during experimental plot studies.

Natural forest buffers

Cooper *et al.* (1987) used ^{137}Cs data and sediment mapping techniques to estimate sediment trapping in a forested buffer zone receiving cropland runoff in a Coastal Plain watershed. The riparian buffer was found to remove 84 to 90% of the sediment eroded from the cropland. Sand and coarse sediments were deposited at the forest edge, while silt- and clay-sized particles were trapped deeper in the buffer. Cooper *et al.* suggest that buffer length should increase as stream order increases because the opportunity for sediment deposition decreases and transport capacity increases as stream order increases. Smith (1989) reported that excluding cattle from a 10 to 13 m length riparian pasture reduced sediment loading to the receiving stream by 87%. Castelle *et al.* (1994) reported that the relationship between buffer length and sediment removal was non-linear in that disproportionately long buffers are required for increased sediment removal. For example, increasing sediment removal from 90 to 95% on a 2% slope would require buffer length to be doubled from 30.5 to 61 m. In a literature review, Gilliam (1994) reported that riparian buffers are the most important factor reducing sediment loadings to receiving waters, with sediment trapping efficiencies of 85 to 90% commonly reported in Coastal Plain regions. In another literature synthesis, Lowrance *et al.* (1995) reported similar trapping efficiencies, 80 to 90%, in forested Coastal Plain buffer zones. Sediment trapping efficiencies in other physiographic regions have not been as well investigated and are probably lower than those reported for the Coastal Plain because of steeper slopes and channelised flow effects.

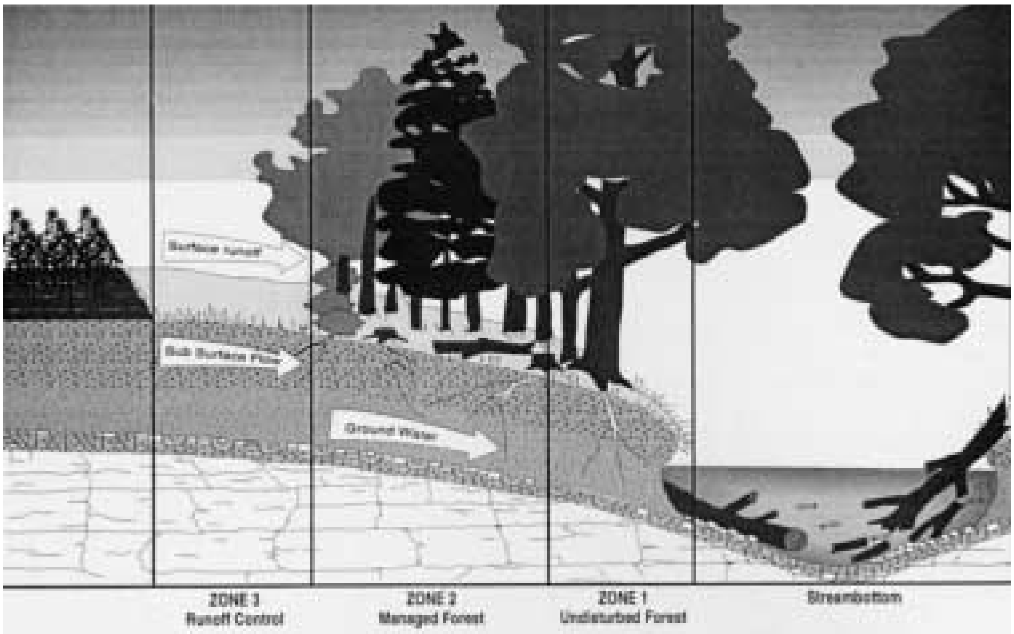
Lowrance *et al.* (1995) reported that buffer zones are the most effective with first and lower order streams because of the greater potential for interaction between upland runoff and the riparian zone in lower order streams. For ephemeral and first order streams, the potential impact of buffer zones in trapping sediment is directly proportional to the proportion of surface runoff from the contributing area that moves through the buffer zone as shallow sheet flow. Smith *et al.* (1993) suggest that buffers are very important for ephemeral channels because they may be greater sources of NPS sediment loads than perennial channels because of their abundance and lower vegetative cover. For second-order and larger streams, sediment reduction will be based on the proportion of surface runoff from the upslope contributing area that flows through the buffer and the proportion of the surface runoff that enters the riparian area through upslope lower order streams. Clearly, as stream order increases, the impact a buffer zone along a particular stream reach can have on the reduction in overall load within that reach is reduced (Lowrance *et al.*, 1995). On a watershed basis, the higher the proportion of streamflow originating from relatively short flow-paths to small streams protected by buffer zones, the greater the potential impact of buffer zones. Similarly, the higher the drainage density of a watershed, the greater the potential benefits of buffer zones.

Zoned riparian buffers

In response to the need for the development of guidelines for the design and management of forest buffer zones, Welsch (1991) suggested that buffer zones should consist of three zones (Figure 1).

Zone 1 is a permanent and undisturbed forested zone immediately adjacent to the stream. Zone 2 is a managed forest zone, just upslope of Zone 1, in which timber is periodically harvested. Zone 3 is a managed herbaceous strip, usually grasses, just upslope of Zone 2 that is used to control runoff. The three-zone forest buffers are specified for habitat and water quality protection of waterbodies adjacent to cropland, pastures and urban areas that are sources of diffuse pollution. Applicable waterbodies included perennial and intermittent stream, lakes, ponds, wetlands and groundwater recharge areas. Required lengths for each of the zones were not presented.

Figure 1. Three zoned riparian buffer zone system (Lowrance *et al.*, 1995)



The principal purpose of the unmanaged forest zone (Zone 1) is aquatic habitat protection and streambank protection. The primary purposes of the managed forest zone (Zone 2) are removal of sediment from overland flow and nutrients from overland and subsurface flow. Zone 3, the managed runoff control zone, has two principal purposes. First, because it is composed of close growing herbaceous vegetation, usually grass, it offers high resistance to overland flow and is thus an effective sediment deposition area under shallow flow conditions. Secondly, because it is a managed and possibly constructed zone, it can be designed to minimise the movement of runoff into Zone 2 as concentrated flow. Under shallow, uniform flow conditions (minimal concentrated flow), Zone 3 will be responsible for most of the sediment trapping in the three zone buffer. Once Zone 3 is inundated with sediment, and/or flow into Zone 2 is no longer shallow and uniform, Zone 3 will have to be reconstructed and trapped sediment moved back up into the contributing source area. Because of known problems with channelised flow through buffer zones, several researchers (Welsch, 1991; Franklin *et al.*, 1992) have suggested that Zone 3 could also incorporate engineered level-lipped spreaders to distributed runoff across a wider portion of Zone 2.

Lowrance *et al.* (1995) provided a comprehensive research synthesis on the likely effectiveness of Welsch's three zone forest buffer system in the Chesapeake Bay drainage basin. They discussed the likely effectiveness of forest buffers for habitat protection and removal of sediment and nutrients in the three different physiographic regions of the Chesapeake Bay. Most of their report deals with the Coastal Plain region where conditions are ideal for buffer zones (widespread existing forested riparian zones, lower slopes, sandy soils and low upland to riparian land ratios, typically 2:1 to 3:1) and where the most detailed research on buffer zones for water quality protection in the US has been conducted. Research on riparian zone effectiveness for sediment removal in the Piedmont and the Valley and

Ridge physiographic provinces was reported to be very limited and it was difficult to make quantitative estimates of buffer zone effectiveness in these areas. Sediment removal by buffer zones in the Piedmont was assumed to be similar but somewhat less than that in the Coastal Plain due to higher slopes and greater channelisation of flow through the buffers. Sediment trapping is assumed lower still in the Valley and Ridge because of higher slopes and increased channelisation but trapping can still be significant if the runoff control zone (Zone 3) is well-maintained and flow channelisation is minimised.

Buffer zones as sediment sources

Although most of the literature views buffer zones as sediment sinks, some researchers have reported that buffers are sometimes sediment sources. In addition, even though buffers are usually net sediment sinks, downstream channel and streambank erosion often negates the benefits of buffer zones with respect to reducing sediment yield. In a Coastal Plain watershed in Maryland, Jordan *et al.* (1993) unexpectedly found that a forested buffer receiving runoff from a no-till corn field was a net sediment source because of active erosion in the channel flowing through the buffer and little opportunity for deposition elsewhere. Erosion in the buffer channel was attributed to higher hydraulic loadings (channelised flow) than reported in other buffer zone studies. The channel erosion in the buffer was also probably due to the low sediment load in the surface runoff from the no-till field. Since the sediment load was low, most of the energy in the runoff was available for sediment detachment and consequently one would expect channel erosion if the channels were not well-protected from scour. Smith (1992) had similar results in a field study in New Zealand on the effects of forested (pine) buffers in reducing NPS pollutant losses from steep pasture land. After eight years of growth, the forested buffers were sediment sources rather than sinks. Sediment losses were 52 to 219% higher from the buffers than from the control pasture. Observations of the soil surface in the buffer zone found that the soil was largely bare or covered with a thin loose layer of pine needles. As the pine canopy had closed in, the original grasses and weeds in the pasture had presumably died leaving the soil with little protection from the erosive scour of the channelised surface runoff from the upslope pastures.

Baker (1992) points out that reductions in soil loss from upland watersheds due to conservation practices often do not result in decreased sediment loadings to higher order streams and lakes. Sediment yields may remain steady despite declining sheet and rill erosion because of increased stream bank and channel erosion required to satisfy the available sediment transport capacity of channelised flow. These phenomena have been observed in classic watershed erosion studies such as Coon Creek, Wisconsin (Trimble, 1981) and are further evidenced by steady suspended solids concentrations at most USGS water quality monitoring stations since the 1970s, despite declining cropland erosion (Baker, 1992).

The above phenomena would be expected in any watershed which experiences a significant decrease in upland erosion due to changes in land management (urbanisation or installation of BMPs for upland erosion control). During the period when upland erosion was high, the downslope buffer areas would be expected to be sediment sinks because of the high incoming sediment load and sediment transport capacity deficit. However, as land use changes and upland erosion decreases (and runoff possibly increases in the case of urbanisation), the erosive potential of runoff entering the buffer would increase due to a sediment transport capacity excess and channel erosion will increase. The buffer and downslope channels would continue to be sediment sources until a new geomorphologic equilibrium in the watershed is reached. This could take years or decades. This effect would be partially offset by decreases in runoff rates and volumes (due to increased infiltration and flow retardance in the buffer zone) that would decrease watershed sediment yield because of reduced sediment transport capacity. Even though watershed sediment yield might not decrease significantly, water quality would almost certainly improve because of the lower nutrient and organic matter content of eroded channel substrate as compared to upland top soils. A possible exception to this would be the erosion and resuspension of high clay content wetland and channel sediments that may be enriched with nutrients and organic matter.

Buffer zone design models

There are very few suggestions or widely accepted methods in the literature that can be used to determine site specific buffer lengths required for a given sediment trapping efficiency. Barfield *et al.* (1979) developed a steady state model, the Kentucky filter strip model, for determining the sediment filtration capacity of grass media as a function of flow, sediment load, particle size, flow duration, slope and media density. Laboratory experiments indicated that clay-sized particles are not trapped in grass buffers, so the model does not allow deposition of clay-sized particles. Outflow concentrations were primarily a function of slope and media spacing for a given flow condition. The Kentucky model was extended for unsteady flow and non-homogeneous sediment by Hayes *et al.* (1979). The model was incorporated into SEDIMOT II (Wilson *et al.*, 1984), an erosion and stormwater management model that is widely used at surface mining sites.

Several attempts have been made to design buffer zones using the CREAMS model (Knisel, 1980). Williams and Nicks (1988) applied CREAMS to a 1.6 ha watershed in Oklahoma. Buffer effectiveness was found to be dependent on buffer zone length, Manning's n, slope and slope shape. Flanagan *et al.* (1989) used the CREAMS model to develop a simplified set of equations that could be used for buffer zone design as a function buffer length, slope and vegetation density (Manning's n). The WEPP model (Flanagan and Livingston, 1995) has also been proposed for use in buffer zone design. WEPP uses more advanced hydraulic and sediment transport routing models than CREAMS, but sediment trapping is still primarily a function of buffer slope and Manning's n. These models, like the Kentucky model, cannot consider the long-term effectiveness of buffer zones because they do not account for sediment accumulations within the buffer zone. Consequently, these models would be expected to overestimate long-term sediment trapping. These models also cannot account for concentrated flow effects that some researchers believe are the most important factor affecting buffer zone performance.

The Riparian Ecosystems Management Model, REMM (Altier, 1994), is a state-of-the-art buffer zone model. The model has sophisticated nutrient and vegetative growth models required for simulating nutrient dynamics in riparian zones, but the simplified hydrologic and sediment transport portions of the model are based on CREAMS era technology and the model's usefulness for estimating sediment dynamics in buffer zones is probably limited.

Prato and Shi (1990) used the AGNPS model to compare the economics of in-field erosion control practices and riparian buffer zones in meeting water quality objectives. Erosion control practices generated greater reductions in sediment yield but were less cost effective than riparian buffer zones in meeting water quality goals. However, in-field erosion control resulted in higher income for farmers than the riparian strategy because of the land taken out of production by the riparian strategy. AGNPS is an event-oriented model that does not simulate the principal long-term processes, such as vegetative growth, that control sediment transport in buffer zones. Consequently, its predictions have limited value for buffer zone design.

Recently, Inamdar (1996) developed a physically-based, continuous simulation, riparian zone model to simulate surface and subsurface runoff and sediment transport in forested and herbaceous buffer zones. Hydrologic portions of the model were successfully validated with field data from Canada, Japan and the US. The sediment transport model was validated using data from Kentucky. Site conditions such as slope gradient, slope shape, flow concentration and soil horizon thickness were shown to play a significant role in shaping hydrologic and sediment transport in buffer zone. Further testing and validation of the model is required, but the model is probably the best model available for predicting long-term sediment dynamics in herbaceous and forested buffer zones.

DISCUSSION

As indicated previously, there is considerable uncertainty and debate regarding the site-specific effectiveness of buffer zones for sediment removal. Until there is a better understanding of surface and subsurface hydrology in riparian areas, it will be impossible to accurately predict the effectiveness of riparian zones in trapping NPS pollutants (Gilliam, 1994). In spite of this uncertainty, there is a general consensus that buffer zones are beneficial for sediment and other pollutant removal and for habitat enhancement. Several models have been developed to help design buffer zones with respect to sediment removal, particularly in grass buffers. Unfortunately, none of these models simulate the effects of sediment accumulations in the buffer and the observed decline in trapping efficiency over time. All of the models assume that buffers are in a virgin state with respect to sediment trapping ability at the beginning of each storm. None of the models except Inamdar's (1996) simulates the effects of concentrated flow on sediment transport. All assume that flow is shallow and uniformly distributed across the width of the buffer. This is a serious limitation since concentration of flow in natural drainageways is the rule rather than the exception in the real world (Dillaha, 1989b). The development of a comprehensive field or watershed scale model for buffer zone design must simulate the effects of natural drainageways upslope of and within the buffer zone and the effects of long-term sediment accumulations to accurately predict buffer zone sediment transport.

Several techniques, including water bars and level-lipped spreaders, were suggested as means to improve buffer trapping efficiency by reducing concentrated flow through buffers. Water bars are probably a more realistic option because they can be quickly constructed using conventional farm implements, require minimal maintenance and do not interfere with other farming operations. Level-lipped spreaders are probably not suitable for agricultural applications because of the high degree of control required in their construction, the expense of construction materials and labour and possibly because of high maintenance requirements. They are probably more suitable for use in urban areas where the costs of alternative pollution control practices are high.

While buffer zones are an essential watershed protection practice, it is critically important in agricultural situations to view them as secondary best management practices (BMPs) to be used only in conjunction with in-field BMPs (Dillaha *et al.*, 1989b; Barling and Moore, 1994). In-field BMPs such as conservation tillage, contouring, strip cropping, controlled grazing, etc. should be given priority over buffer zones for NPS pollution control because they prevent pollutant generation at its source. Sediment and nutrients are valuable natural resources when they are in the field. They only become pollutants when they leave the field. Buffer zones trap pollutants downslope on the field and they are lost from the agricultural production system unless they are collected and hauled back to the fields. Hauling sediment back to the fields is not economically feasible in a forest buffer and difficult in herbaceous buffers as traditional farm machinery is not suited for moving soil more than a few metres. While in-field BMPs are preferred for high yield agricultural production systems, in-field BMPs alone do not provide adequate off-site water quality protection. In-field BMPs may hold sediment losses to acceptable levels, but current BMPs have a difficult time controlling agricultural chemical losses. Consequently, a combination of buffer zones and in-field BMPs is required if we are to have both high yield agriculture and improved water resources (Karr and Schlosser, 1978).

ACKNOWLEDGEMENTS

The buffer zone research conducted by the authors of this paper was supported by the Virginia Water Resources Research Centre, the USEPA, the Virginia Agricultural Experiment Station and USDA Cooperative Regional Project S-249.

REFERENCES

Altier, L.S. (1994) *REMM: Riparian Ecosystem Management Model.* Unpublished User's Manual. Southeast Watershed Research Laboratory, USDA-ARS, Tifton, GA.

Baker, L.A. (1992) Introduction to non-point source pollution in the United States and prospects for wetland use. *Ecological Engineering,* **1**, 1-26.

Barfield, B.J., Tollner, E.W. and Hayes, J.C. (1979) Filtration of sediment by simulated vegetation, I. Steady-state flow with homogeneous sediment. *Transactions of the ASAE,* **22**(3), 540-545, 548.

Barling, R.D. and Moore, I.D. (1994) Role of buffer strips in management of waterway pollution. *Environmental Management*, **18**(4), 543-558.
Castelle, A.J., Johnson, A.W. and Conolly, C. (1994) Wetland and stream buffer size requirements. *J. of Environmental Quality*, **23**, 878-882.
Clark, E.H. (1985) The off-site costs of soil erosion. *J. Soil and Water Conservation*, **40**, 19-22.
Cooper, J.R., Gilliam, J.W., Daniels, R.B. and Robarge, W.P. (1987) Riparian areas as filters for agricultural sediment. *Soil Sci. Soc. Am. J.*, **51**(2), 416-420.
Dillaha, T.A., Reneau, R.B., Mostaghimi, S. and Lee, D. (1989a) Vegetative filter strips for agricultural non-point source pollution control. *Transactions of the ASAE*, **32**(2), 491-496.
Dillaha, T.A., Sherrard, J.H. and Lee, D. (1989b) Long-term effectiveness and maintenance of vegetative filter strips. *Water Environment and Technology*, **1**(3), 418-421.
Flanagan, D.C., Foster, G.R., Neibling, W.H. and Burt, J.P. (1989) Simplified equations for filter strip design. *Transactions of the ASAE*, **32**, 2001-2007.
Flanagan, D.C. and Livingston, S.J. (1995) *USDA-Water erosion prediction project (WEPP):* technical documentation. NSERL Report No. 10. National Soil Erosion Research Lab., West Lafayette, IN.
Franklin, E.C., Gregory, J.D. and Smolen, M.D. (1992) *Enhancement of the effectiveness of forest filter zones by dispersion of agricultural runoff.* Report No. UNC-WRRI-92-270. Water Resources Research Institute, Raleigh, NC.
Gilliam, J.W. (1994) Riparian wetlands and water quality. *J. of Environmental Quality*, **23**(5), 896-900.
Hayes, J.C., Barfield, B.J. and Barnhisel, R.I. (1979) Filtration of sediment by simulated vegetation II. Unsteady flow with non-homogeneous sediment. *Transactions of the ASAE*, **22**(5), 1063-1067.
Inamdar, S.P. (1996) *Investigation of hydrologic and sediment transport processes on riparian hillslopes.* Ph.D. Dissertation. Virginia Polytechnic Institute and State University, Blacksburg.
Jordan, T.E., Correll, D.L. and Weller, D.E. (1993) Nutrient interception by a riparian forest receiving inputs from adjacent cropland. *J. of Environmental Quality*, **22**(3), 467-473.
Karr, J.R. and Schlosser, I.J. (1978) Water resources and the land-water interface. *Science*, **201**, 229-234.
Knisel, W.G. (Ed.) (1980) *CREAMS: A field-scale model for chemical, runoff and erosion from agricultural management systems.* Conservation Research Report No. 26. U.S. Dept. Agric., Washington, D.C. pp 640.
Lowrance *et al.* (1995) *Water quality functions of riparian forest buffer systems in the Chesapeake Bay Watershed.* Report No. EPA 903-R-95-004. U.S. Environmental Protection Agency, Washington, D.C.
Magette, W.L., Brinsfield, R.B., Palmer, R.E. and Wood, J.D. (1989) Nutrient and sediment removal by vegetated filter strips. *Transactions of the ASAE*, **32**(2), 663-667.
Neibling, W.H. and Alberts, E.E. (1979) *Composition and yield of soil particles transported through sod strips.* ASAE Paper No. 79-2065. Am. Soc. of Agric. Engrs., St. Joseph, Mich.
Prato, T. and Shi, H. (1990) A comparison of erosion and water pollution control strategies for an agricultural watershed. *Water Resources Research*, **26**(2), 199-205.
Smith, C.M. (1989) Riparian pasture retirement effects on sediment, phosphorus and nitrogen in channelised surface runoff from pastures. *New Zealand J. of Marine and Freshwater Resources*, **23**, 139-146.
Smith, C.M. (1992) Riparian afforestation effects on water yields and water quality in pasture catchments. *J. of Environmental Quality*, **21**, 237-245.
Smith, M.A., Dodd, J.L., Skinner, Q.D. and Rogers, J.D. (1993) Dynamics of vegetation along and adjacent to an ephemeral channel. *J. of Range Management*, **46**(1), 56-64.
Tollner, E.W., Barfield, B.J. and Hayes, J.C. (1982) Sedimentology of erect vegetal filters. *Proceedings of the Hydraulics Division, ASCE*, **108**(12), 1518-1531.
Trimble, S.W. (1981) Changes in sediment storage in the Coon Creek basin, Driftless Area, Wisconsin, 1853-1975. *Science*, **214**:, 181-183.
USEPA (1995) *Water quality conditions in the United States: A profile from the 1994 national water quality inventory report to Congress.* Publication EPA841-F-95-010. Office of Water. Washington, D.C.
Welsch, D. (1991) *Riparian forest buffers – function and design for protection and enhancement of water resources.* USDA-FS Pub. No. NA-PR-07-91. USDA-FS, Radnor, PA.
Williams, R.D. and Nicks, A.D. (1988) Using CREAMS to simulate filter strip effectiveness in erosion control. *J. of Soil and Water Conservation*, **43**(1), 108-112.
Wilson, L.G. (1967) Sediment removal from flood water by grass filtration. *Transactions of the ASAE*, **19**(1), 35-37.
Wilson, B.N., Barfield, B.J. and Moore, I.D. (1984) A hydrology and sedimentology watershed model. Part I: Modeling techniques. In: *Design manual for the SEDIMOT II hydrology and sedimentology model.* Dept. of Agricultural Engineering, University of Kentucky, Lexington.
Young, R.A., Huntrods, T. and Anderson, W. (1980) Effectiveness of vegetative buffer strips in controlling pollution from feedlot runoff. *J. Environ. Qual.*, **9**, 483-487.

The interactions of buffer zones and phosphorus runoff

5

J. Uusi-Kämppä[1], E. Turtola[2], H. Hartikainen[3] and T. Yläranta[1]

[1] *Agricultural Research Centre of Finland, Institute of Soils and Environment, FIN – 31600, Jokoinen, Finland*
[2] *Agricultural Research Centre of Finland, Institute of Crop and Soil Science, Finland*
[3] *University of Helsinki, Department of Applied Chemistry and Microbiology, Finland*

Abstract

Phosphorus (P) is transported in surface runoff from a drainage area to watercourses in dissolved (DP) and particulate (PP) fractions. To alleviate eutrophication of the recipient water bodies, P losses should be diminished by minimising P dissolution and the detachment of P-carrying soil particles, and by measures removing P from the surface runoff water. In certain areas, buffer zones – i.e. buffer strips, ponds and wetlands – may be effective traps for P. Vegetated buffer strips between fields and recipient water bodies remove PP through deposition of suspended particles from the runoff water on the soil surface and from the water infiltrating into soil pores. Small ponds and artificial wetlands retain PP through sedimentation of particles by decreasing the water flow velocity. DP concentration in runoff may be decreased in buffer strips through sorption by soil components, especially during infiltration, and through biological uptake by plants and microorganisms. In ponds and wetlands, removal of DP depends on adsorption by the suspended solids and bottom deposits, and microbial and plant uptake. Sometimes the concentration of DP may increase during transport through buffer zones, due to the release of P from decaying vegetation. Moreover, rooted macrophytes pump P from the pond sediment and release it in dissolved form. During transport through buffer zones, the sorption/desorption equilibrium between dissolved phosphorus, suspended sediment and soil is affected by the properties of the solid phase and ambient solution. The sorption capacity of the solid phase is dependent on the content of Al and Fe oxides, organic matter and calcium carbonate. The desorption tendency, by contrast, is markedly dependent on the degree of P saturation on the oxide surfaces. Therefore, in the source soil, P losses can be effectively diminished by reducing the accumulation of fertiliser P. Apart from the amount of sorptive agents in the buffer zones, the efficiency of strips and ponds as P traps is affected by kinetic factors, contact time and temperature. The best buffer zones can retain half of the P from agricultural runoff. They should be effective in field areas which are most susceptible to runoff and erosion. The highest retention of sediment and P is achieved in small watersheds, close to the point of erosion. Compared with PP, the retention of DP is much less efficient, and in some cases the losses may even be increased.

PRINCIPAL SOURCES OF P WITHIN CATCHMENTS

Phosphorus is carried to watercourses from point sources such as industrial or municipal wastewater and from diffuse sources such as forested or agricultural areas. Point sources are easy to identify and control, and they have been reduced since the late 1960s. As further point-source control becomes less cost-effective, attention is now being directed towards the contribution of diffuse P load from agricultural soils. In Scandinavia, the contribution of agriculture to the total loading of P is estimated to be 39% in Denmark, 54% in Norway, 73% in Sweden and 79% in Finland (Löfgren and Olsson, 1990; Kronvang and Svendsen, 1991; Græsbøll *et al.*, 1994; Rekolainen *et al.*, 1995).

The different P forms in runoff are often operationally grouped into dissolved (DP) and particulate (PP) fractions, according to the method of determination and potential availability to algae. PP consists of undissolved inorganic and organic compounds in colloids and larger particles. According to bioassay experiments, 5 to 41% of the PP in agricultural runoff is biologically available (Persson, 1990;

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0 © 1997 Quest Environmental.

Ekholm, 1994). As for point sources, the availability has been found to be higher: 30% in urban runoff (Cowen and Lee, 1976), 40 to over 90% in purified wastewater (Young *et al.,* 1982; Shannon, 1983; Berge and Källqvist, 1990; Persson 1990) and 80% in the effluent of a pulp and paper mill (Priha, 1994).

In surface runoff, the proportion of P in dissolved form is reported to vary from 17 to 45% (Pietiläinen and Rekolainen, 1991). In contrast too much PP, dissolved orthophosphate P (PO_4-P), the main form of inorganic dissolved P in runoff waters (Broberg and Persson, 1988), is considered directly available to algae (Walton and Lee, 1972; Ekholm *et al.,* 1991; Ekholm, 1994).

PP and DP losses from soil are induced by quite different mechanisms: erosion producing suspended soil particles and thus PP to runoff, and desorption of DP both from the soil remaining in the field and from the suspended soil particles (e.g. Logan, 1982; Yli-Halla *et al.,* 1995). In general, the P content of the eroded particulate material is greater than that of the source soil, due to preferential transport of clay-sized particles rich in sorptive oxides. The transport of DP in runoff is initiated by desorption, dissolution and extraction of P from soil and plant material. These processes occur as a portion of rainfall or snowmelt water interacts with a thin layer of surface soil before leaving the field as runoff (Sharpley, 1985).

The P buffering by the soil means that, depending on the circumstances, phosphate anions can be sorbed to, or desorbed from, the solid particles. For each soil-solution system, there is an equilibrium phosphate concentration in the solution, above which P is sorbed and below which P is desorbed to the solution (Beckett and White, 1964; Hartikainen, 1991). As mineral soils often retain P effectively, a progressive build-up of residual fertiliser P has resulted in a larger P content in the surface layer of soils compared with the subsoil. As a consequence, the P sorption capacity of surface soils may become slowly saturated, and the soil surface will support a higher concentration of DP in runoff. In the case of deep infiltration and flow to the drainage system, however, the slow movement of water through the subsoil may allow sorption of DP from the percolating waters (Hartikainen, 1979; Sharpley and Syers, 1979; Turtola and Paajanen, 1995). Therefore, in most cases, P export from catchments occurs in surface rather than subsurface runoff.

Within a watershed, the prevailing cropping practices influence the magnitude of P losses and the relative abundance of PP and DP. Reflecting the extent of soil erosion, PP losses tend to be highest from bare fallow, high from row crops, moderate from cereals and fertilised leys and lowest from perennial unfertilised leys (e.g. Gustafson, 1982; Brink *et al.,* 1987; Turtola, 1994). However, soil erosion may be high from heavily grazed pastures (Heathwaite *et al.,* 1990) thus promoting large PP losses. Due to lower PP losses from ungrazed leys the proportion of DP in runoff tends to be higher from these areas (Uhlen, 1978; 1988). Large DP losses have been measured from areas where fertiliser P is broadcast on the soil surface rather than incorporated (Timmons *et al.,* 1973; Sharpley *et al.,* 1992; Turtola and Jaakkola, 1995).

RECOMMENDATIONS FOR ESTABLISHING BUFFER ZONES

A few general recommendations exist for establishing buffer zones. The width of the buffer strips depends on the size of the source area of the runoff. Narrow buffer strips may be sufficiently wide between small field areas and open ditches and brooks. However, wider strips or zones are needed on fields with steep slopes along watercourses (Ahola, 1990).

Some computer models e.g. GRAPH (Lee *et al.,* 1989) and CREAMS (Chemical, Runoff, Erosion and Agricultural Management Systems, reported by Williams *et al.,* 1990), may be used to simulate P transport in grass buffer strips. It is also possible to use these models to provide recommendations for the width of buffer strips.

Within a watershed, small ponds and wetlands are recommended near P sources. The lower reaches of watersheds are considered to be natural sites for large wetlands (Hammer, 1992). The ponds and wetlands should be large enough for sedimentation of small particles even during peak flows. The ratio of pond area to the area of the whole drainage basin should be at least 1:1000. To retain soluble nutrients as well, the ratio should be 1:50. The slopes around the pond and wetland must also be shallow enough to reduce bank erosion.

In the United States, a combination of a vegetative filter strip and a riparian ecosystem has been recommended to provide both short- and long-term control of agricultural pollutants (Lowrance, 1991). The system comprises three zones of vegetation between field and stream. Zone 1, nearest to the stream, is a narrow zone (5-15 metres) of woody vegetation to provide streambank stabilisation and shading of the stream. Zone 2 is wide enough to remove nutrients associated with eroded soil particles to background levels. Zone 3, upslope from Zone 2 is a vegetative buffer strip designed primarily for coarse sediment deposition.

In Finland, the establishment of buffer zones is supported by the Government and the EU. The rules for buffer strips require a strip of one metre on the sides of main ditches, and a strip of three metres, covered by perennial vegetation, on the sides of brooks and watercourses, rivers, lakes and the sea, and around household wells. In addition, the establishment of wider buffer strips, ponds and wetlands may be eligible for financial support. Buffers with a width of over 15 metres are recommended on the sides of brooks and watercourses, provided that they are left uncultivated for 20 years. According to the recommendations, the area of the pond/wetland should be at least 0.15% of the whole drainage area and 0.2% of the field area. The financial support for buffer strips, ponds and wetlands is (max) 3600 FIM ha^{-1} (790 US$ ha^{-1}).

EXPERIMENTAL RESULTS CONCERNING THE EFFICIENCY OF BUFFER ZONES AS P TRAPS

According to experimental studies, the efficiency of buffer strips, ponds and wetlands in retaining P varies considerably (Tables 1 and 2).

Table 1. Retention (%) of Total Phosphorus (Tot-P) and Orthophosphate Phosphorus (PO_4-P) in Buffer Strip Studies.

Author	*Location*	*Source Area (ha)*	*Buffer Width (m)*	*Retention (%) Tot-P*	*PO_4-P*	*Comments*
Dillaha *et al.* (1989)	Virginia, USA	0.01	4.6	49-85	69-83	Simulated rainfall
			9.1	65-93	48-31	Increase of PO_4-P
Magette *et al.* (1987)	Maryland, USA	0.01	4.6	41		Less effective over time
			9.2	53		
Syversen (1995)	Norway	0.045	5	45-56	2-77	Natural rainfall,
			10	56-85	0-88	slope of 12-17%, and
			15	73	10	strips with native grass
Uusi–Kämppä and Yläranta (1996)	Southern Finland	0.063	10	20-36(a)	0-62	Natural rainfall, increase of PO_4-P
Uusi–Kämppä (unpublished data)	Southern Finland	0.063	10	53-78(a)	33-33	During summer 1995
Schwer and Clausen (1989)	Vermont, USA	wastewater	26	89	92	Greatest removal in growing season
Vought *et al.* (1994)	Sweden		8		66	Greatest removal within first metres
			16		95	

(a) = Particle-bound P

Table 2. Retention (%) of Total Phosphorus (Tot-P) and Orthophosphate Phosphorus (PO_4-P) in Ponds and Wetlands.

Author	*Location*	*Source of P*	*Pond Area (ha)*	*Retention (%) Tot-P*	*PO_4-P*	*Comments*
Sedimentation pond						
Brown *et al.* (1981)	Idaho, USA	Runoff water	0.09	25-33		Efficiency depended on flow rate
Hirvonen *et al*. (1996)	Southern Finland	d.a. of 580 ha	0.18	35		No removal after two years
Lindkvist (1992)	Southern Sweden	d.a. of 650 ha		8	-8	In autumn, increase of PO_4-P
Lindkvist and Håkansson (1993)	Southern Sweden	d.a. of 650 ha		22	-19	In spring, increase of PO_4-P
Submerged macrophyte pond						
Hörberg and Kylefors (1991)	Southern Sweden	w.w.	1.1	62		Annual P retention 0.03 g/m^2d
Mander *et al.* (1991)	Estonia	w.w. from a farm		56-67		Greatest removal in growing season
Bioditch						
Mander *et al.* (1991)	Estonia	w.w. from a farm		60-92		Greatest removal in growing season
Root-zone						
Hörberg and Kylefors (1991)	Southern Sweden	w.w.	0.11	61		Annual P retention 0.4 g/m^2d
Mander *et al*. (1991)	Estonia	w.w. from a farm		60-75		Greatest removal in growing season
Artificial Wetland						
Braskerud (1994)	Southern Norway	d.a. of 50-100 ha	0.02-0.09	20-42		Greatest removal in winter
Jenssen *et al.* (1993, 1995)	Southern Norway	w.w.	0.01	98		A combination of four units
Natural Wetland						
Gehrels and Mulamoottil (1990)	Canada	Runoff water	18		-22	Increase of PO_4-P
Mander *et al.* (1991)	Estonia	w.w.	18	27-88		Greatest removal in summer
Pommel and Dorioz (1995)	Switzerland / France	Runoff water	3	65	65	During stormflows and lowflows

d.a. = drainage area
w.w. = wastewater

In Norway, Syversen (1994) studied the efficiency of filter strips with widths of 5, 10 and 15 metres planted with native vegetation. More than 50% of the incoming P and 0-88% of DP was removed. In terms of P retention, there were no differences between strips covered with forest and grass. Mander *et al.* (1991) reported alder forests and willow brushes to be the most effective biotopes for buffer strips. Ten-metre-wide strips were able to adsorb/transform almost 100% of the incoming P.

In buffer strips, the filtration process is found to be of an exponential nature: on the upper part of strips the amount of adsorbed and transformed P is significantly larger than that in the middle and lower part of the buffer (Mander *et al.,* 1991; Vought *et al.*, 1994). Thus even narrow forest/brush strips may be important in removing nutrients from runoff water.

P retained in buffer strips may be transformed into more mobile forms which may subsequently be lost into an adjacent water body. In an artificially-created rainfall situation, Dillaha *et al.* (1988) found that buffer strips may increase DP losses. Uusi-Kämppä and Yläranta (1996) obtained similar results from buffer strip plots with native vegetation: the loss of DP was over 50% greater from plots with native vegetation as compared to plots with no buffer strip or with a grass buffer strip. High losses of DP from buffer strip plots with native vegetation may have been due to the release of P from decaying grass residue in the spring. In a model simulation, Lee *et al.* (1989) found that above-ground biomass in a buffer strip was a more significant source of P than the source soil.

According to several studies, the retention of DP is often rather low (e.g. Dillaha *et al.*, 1988; Lee *et al.*, 1989; Uusi-Kämppä and Yläranta, 1996; Williams *et al.*, 1990). Moreover, a buffer zone may at first act as an efficient sink for P and then turn to a source of P (Richardson, 1985; Dillaha *et al.*, 1988, 1989; Magette *et al.*, 1989; Vanek, 1991). This may happen when environmental conditions change, when the soil material becomes gradually enriched with P or when the assimilative capacity of a buffer zone is exhausted, allowing previously trapped P to be released from the filter vegetation and soil as DP. Dillaha *et al.* (1988) assumed that DP removal should decrease with time as filtration decreases, the adsorption capacity of vegetation is saturated and surface soil P sorption sites become occupied.

Both natural and artificial wetlands have been used for removing P from domestic wastewater and agricultural runoff (e.g. Richardson, 1985; Mander *et al.*, 1991; Baker, 1992; Mitch, 1992; Braskerud, 1994; Jenssen *et al.*, 1995). Artificial wetland ecosystems with aquatic vascular plants, vegetated bioponds and bioditches on natural soil, root zone systems in water bodies, and vegetated riparian buffer strips were able to adsorb and/or transform P (Mander *et al.*, 1991). In Norway, an artificial wetland system planted with Phragmites and Typha and modified with a fabricated porous medium (Leca 0-4 mm) with high P adsorption capacity showed an average removal of 97% for P over a period of 18 months (Jenssen *et al.*, 1993, 1994).

Wetlands may convert some PP to plant available PO_4-P, thus contributing to downstream eutrophication problems (e.g. Gehrels and Mulamoottil, 1990). This conversion is considered to be due to the intense leaching of decaying vegetation and high water levels, which induces anaerobic conditions, thus increasing the solubility of phosphate.

In Finland, Hirvonen (1994) reported on an experimental pond, built in a ditch to retain P from agricultural field waters. The retention of P was best in the first year, when as much as two thirds of the incoming P was trapped. In the second year only one fifth of the P was retained, and over the last two years the pond did not retain any P (Hirvonen *et al.*, 1996). Richardson (1985) reported that wetlands used for wastewater filtration became P-saturated in just a few years.

Wetlands may temporarily act as a sink or a source for P. In summer, the wetlands usually retain P because of sedimentation. In autumn, however, the high flows resuspend the sediment and carry it forward in the watercourse. Therefore, P retained during the summer period can be a major source of nutrient transport in the autumn (Svendsen, 1992; Taponen, 1995). Moreover, the water-sediment interface may become anoxic also during summer, increasing the release of PO_4-P.

PRINCIPAL PROCESSES ACCOUNTING FOR THE RETENTION OF P WITHIN BUFFER ZONES

Phosphorus retention and removal from agricultural runoff in buffer zones is driven by a combination of chemical, biological and physical processes. The most important mechanisms are deposition of P with sedimenting material, sorption of DP by soil and uptake of P by vegetation (Dillaha *et al.*, 1988; Hörberg and Kylefors, 1991; Lowrance, 1991). In addition, infiltration of runoff water into the soil profile and filtration of suspended solids by vegetation cause the retention of P from runoff water (Dillaha *et al.*, 1988).

Sedimentation and infiltration

Dense vegetation on a buffer strip increases the hydraulic roughness, decreasing overland flow velocity and sediment transport capacity. Sedimentation of particles is a major process removing P from cropland runoff, both in vegetated buffer strips and in wetlands (Dillaha *et al.*, 1989; Baker, 1992). In many cases, however, only the large sand-sized and silt-sized particles and aggregates are deposited (Brown *et al.*, 1981). A large portion of the PP may be bound to particles which are too small to settle out in small ponds and wetlands (Brown *et al.*, 1981).

The ability of ponds to remove PP is controlled by stream power. High flow rates, outside the growing period especially, may overload a small pond, resulting in an increase in velocities to a point where even silt-sized particles may not settle. During low flows, silt-sized particles may settle, but the clay-sized particles cannot be trapped without the aid of flocculation (Brown *et al.,* 1981; Taponen, 1995). Furthermore, in vegetated ponds and wetlands, plant stems and leaves obstruct flow and thus facilitate sedimentation (Hammer, 1992).

In buffer strips, infiltration of runoff water may facilitate reduction of both PP associated with small particles and of DP. During infiltration PP may be sieved from the water in the soil profile (Dillaha *et al.,* 1988). Vegetation may also change the soil structure by creating root channels and thereby increasing the infiltration capacity (Vought *et al.,* 1994). Furthermore, infiltration into the buffer strip soil decreases surface runoff, which in turn reduces the ability of the runoff water to transport soil particles and PP (Dillaha *et al.,* 1988).

Adsorption

The action and efficiency of buffer zones in reducing the DP load from cultivated soils is dependent on the dynamic equilibrium between soil and dissolved P. The direction of the net reaction depends on the sorptive properties of the solid material and on environmental factors such as the P and salt concentrations in the runoff water, solution to soil ratio and temperature.

The sorptive reaction is initially fast but continues slowly for many days. The slow phase is attributable to the diffusion of P into the porous oxide material, which renders P less desorbable. Therefore, the contact time is also of significance in controlling the load of DP. It is obvious that during fast flow over a narrow strip of buffer zone soil, the sorption reactions do not have time to occur.

The reduction of the DP by the buffer zones is markedly dependent on the pathway of the runoff water (Burt, this volume). The sorptive properties can be utilised most efficiently in the case of high infiltration capacity and water passing through the soil mass in the buffer zone.

Assimilation by plants

During the growth season, P is retained from runoff water by plant uptake and incorporation of P into biomass. However, in a cold climate the P assimilation is reduced (e.g. Granéli, 1990; Mander *et al.,* 1991), especially during the spring runoff period.

The amount of P assimilation varies between plant species. The incorporation of nutrients into tree biomass seems to be an important sink for P (Peterjohn and Correll, 1984; Leonardsson, 1994). Uusi-Kämppä and Yläranta (1996) observed that native herbage vegetation on buffer strips took up more P than grass stands of timothy and meadow fescue. Hörberg and Kylefors (1991) observed that in a pond with Canadian waterweed *(Elodea Canadensis)* and *Cladophora glomerata,* and in a root-zone with common reed *(Phragmites australis),* the retention of P was 60% from wastewater. However, harvested biomass contained only about 16% of the retained P. In a study by Schwer and Clausen (1989), grass vegetation on buffer strips took up only 2.5% of the incoming P. Some systems with periodic plant harvesting slightly increased direct removals by plants (Granéli, 1990; Hammer, 1992; Osborne and Kovacic, 1993). In wetlands, the plants generally take up only very small quantities (<5%) of the nutrients or other substances present in the influent waters (Hammer, 1992).

Rooted emergent wetland vegetation may sometimes act as a source of P. The vegetation takes up P in summer, but after tissue death the plant P is released in autumn. Thus, vegetation serves only as a short-term sink for P unless the biomass is harvested. In addition, bottom-feeding fish (e.g. carp, catfish) and birds may cause resuspension and thus increase P concentration in wetland outlet water.

EFFECT OF THE INTERNAL PROPERTIES OF THE BUFFER ZONE SOIL ON P RETENTION

In buffer strips, the extent of the retention of soluble P is determined largely by the amount of sorptive components in the soil. In acid soils, P sorption is considered to occur mainly by a specific ligand exchange on to the hydrated Fe and Al oxides. Other soil components capable of retaining P are Fe^{3+} and Al^{3+} ions associated with humic compounds, and some clay minerals, especially allophane. In calcareous soils, the concentration of P is also controlled by the solubility of Ca-phosphate compounds.

The extent of sorption depends on pH and the saturation degree of the oxide surface. At a high pH, the OH- ligands on the oxide surfaces tend to dominate over H_2O-groups, rendering the particle surfaces negative. With decreasing pH the OH- groups protonate to form H_2O ligands, and the negative charge decreases thus facilitating P sorption. Even if the ligand exchange takes place on the oxide surface, irrespective of the charge, it is favoured by a low pH. This response can be explained by two factors: 1) owing to the decrease in negative charge the anion repulsion diminishes, and 2) the H_2O ligands formed are more easily replaced by phosphate than OH- ligands.

The sorption affinity of oxide surfaces is known to decrease with increasing P coverage. This is not only attributable to the fact that the number of sorption sites available diminishes: another factor is that sorption of phosphate anions can render the oxide surface more negative and thus disfavour further sorption. Similarly, other anions possessing the same specific sorption mechanism on to the oxide surfaces can retard or reduce the retention of P through competition. Organic anions, especially humic compounds, can effectively block sorption sites and lower the P retention in surface soils.

EFFECT OF ENVIRONMENTAL FACTORS ON P RETENTION

Some external factors contribute to the retention of DP from the runoff water passing over the surface or infiltrating into the buffer strip soil. A low salt concentration in the solution favours the desorption of P from the suspended soil particles and from the bulk soil mass to the surface runoff water. A decrease in the ionic strength of the solution is shown to markedly enhance the desorption or, *vice versa,* to depress the sorption (e.g. Yli-Halla and Hartikainen, 1996). Similarly, an increase in the solution:soil ratio during surface runoff is capable of favouring the desorption by diluting the salt concentration in the solution phase. As a result of the P buffering, a dilution-induced decrease in the dissolved P concentration is counteracted by P release from the soil material. Furthermore, oxygen depletion usually results in an increased P concentration in the solution. This response can be attributed to the reduction of Fe^{3+} in the oxyhydroxides to Fe^{2+}, whose affinity for P retention is lower. A contributing factor is the tendency of pH to increase under reduced conditions, which in turn promotes the P desorption. However, it is unlikely that the efficiency of the buffer strips is limited by low redox potential. In the ponds, by contrast, oxygen depletion may occasionally appear and induce some desorption of P from the bottom material or the suspended particles.

SEASONAL VARIATION IN THE EFFICIENCY OF BUFFER ZONES

A large seasonal variation in temperature may contribute somewhat to P exchange within the buffer zone. It has been shown that, depending on the P concentration in solution, an increase in temperature can promote P release from the soil as well as the retention of P by the soil (Yli-Halla and Hartikainen, 1996). This response means that the higher the temperature is, the more effectively a high P concentration in the solution is diminished through sorption.

In the Nordic region, the temperature of runoff water during the peak flows in autumn and particularly in spring is much lower (between 0 and 5°C) than temperatures prevailing during the summer runoffs. In buffer strips, the low temperatures of soil and runoff will decrease the retention of DP compared with conditions prevailing in summer. Moreover, the marked seasonal variation in other external conditions, such as ionic strength and volume of runoff as discussed above, will affect the sorption reactions in the buffer zones too.

The efficiency of buffer zones may be considerably affected by snow cover and soil frost (Schwer and Clausen, 1989). Depending on the moisture content during freezing, soil pores may be blocked by ice.

As a consequence, the lower infiltration capacity in the soil surface will decrease the ability of the buffer zone soil to trap PP and DP.

CONCLUSIONS

Obviously, the most efficient approach to reducing P losses from agriculture employs a combination of (1) the best management practices to reduce P losses from the source soils, including management to improve water infiltration for diminishing surface runoff and (2) the establishment of buffer zones to further decrease the P load entering the waterways.

The effect of buffer strips to trap PP is mostly attributable to reduction of waterflow velocity. As buffer zones, the lower ends of concave slopes, where waterflow velocity is markedly reduced, could act as sedimentation areas for incoming PP. In contrast, buffer strips established at the lower ends of convex slopes would primarily reduce losses of PP only from that particular slope area. For maximum P retention efficiency, the buffer zone soil should have high porosity to enable infiltration of large amounts of water. Moreover, a dense, native vegetation with high species diversity and deep-rooted plants would best promote trapping of incoming P by plants. Buffer zones are able to decrease quite efficiently the load of particle-bound P, which is the main form of P in agricultural runoff, though not necessarily the main source of available P. Therefore, buffer strips and sedimentation ponds should be established at the lower ends of fields with high erosive cultivation producing relatively large losses of PP. In particular, depressions which channel water from large areas should be considered as potential sites for these zones.

Compared with PP, the retention of DP in buffer zones is much less efficient, and in some cases the losses of DP are even increased. Moreover, though the buffer zones are efficient initially in trapping P, this may decrease over time.

Drawing a conclusion from the processes of P retention in the buffer zones discussed above, one may state that an acidic soil rich in oxide material and clay and low in adsorbed P and organic matter would best serve as a chemical sink for P. These requirements are usually fulfilled by a fine-textured subsoil material, provided that the permeability is high enough to allow infiltration. It should be taken into account, however, that the accumulation of erosion material can change the mineralogical, chemical and physical properties of the buffer zone soil.

REFERENCES

Ahola, H. (1990) Vegetated buffer zone examinations on the Vantaa River basin. *Aqua Fennica,* **20**, 65-69.

Baker, L.A. (1992) Introduction to nonpoint source pollution in the United States and prospects for wetland use. *Ecological Engineering,* **1**, 1-26.

Beckett, P.H.T. and White, R.E. (1964) Studies on the phosphate potentials of soils. III. The pool of labile inorganic phosphate. *Plant and Soil,* **21**, 253-282.

Berge, D. and Källqvist, T. (1990) *Biotilgjengelighet av fosfor i jordbruksavrenning.* Sammenliknet med andre forurensningskilder, Sluttrapport. Niva, Oslo. pp. 130.

Braskerud, B. (1994) Sedimentation of phosphorus and soil particles in constructed wetlands. In: Persson, R. (Ed.) *Agrohydrology and Nutrient Balances.* Proceedings of NJF-seminar 247, Divison of Agricultural Hydrotechnics, Swedish University of Agricultural Sciences, Uppsala, Sweden. Communications 94, 5, pp. 66-69.

Brink, N., Kreuger, J. and Torstensson, G. (1987) Nutrient losses from arable land. (In Swedish with English summary.) *Ekohydrologi,* **25**, 41-50.

Broberg, O. and Persson, G. (1988) Particulate and dissolved phosphorus forms in freshwater: composition and analysis. *Hydrobiologia,* **170**, 61-90.

Brown, M.J., Bondurant, J.A. and Brockway, C.E. (1981) Ponding surface drainage water for sediment and phosphorus removal. *Transactions of the American Society of Agricultural Engineers,* **25**, 1478-1481.

Cowen, W.F. and Lee, G.F. (1976) *Algal nutrient availability and limitation in Lake Ontario during IFYGL.* Part 1. Available phosphorus in urban runoff and Lake Ontario tributary waters. U.S. Environmental Protection Agency, Environmental Research Laboratory, Duluth, Minnesota. EPA-600/3-76-94a.

Dillaha, T.A., Sherrard, J.H., Lee, D., Mostaghimi, S. and Shanholtz, V.O. (1988) Evaluation of vegetative filter strips as a best management practice for feed lots. *Journal of the Water Pollution Control Federation,* **60**, 1231-1238.

Dillaha, T.A., Reneau, R.B., Mostaghimi, S. and Lee, D. (1989) Vegetative filter strips for agricultural nonpoint source pollution control. *Transactions of the American Society of Agricultural Engineers,* **32**, 513-519.

Ekholm, P. (1994) Bioavailability of phosphorus in agriculturally loaded rivers in southern Finland. *Hydrobiologia,* **287**, 179-194.

Ekholm, P., Yli-Halla, M. and Kylmälä, P. (1991) Availability of phosphorus in suspended sediments estimated by chemicalextraction and bioassay. *Verhandlungen der Internationale Vereinigung für theoretische und angewandte Limnologie,* **24**, 2994-2998.

Gehrels, J. and Mulamoottil, G. (1990) Hydrologic processes in a southern Ontario wetland. *Hydrobiologia,* **208**, 221-234.

Granéli, W. (1990) Standing crop and mineral content of reed, *Phragmites australis* (CAV.) TRIN. ex STEUDEL, in Sweden – Management of reed stands to maximize harvestable biomass. *Folia Geobotanica et Phytotaxonomica,* **25**, 291-302.

Græsbøll, P., Erfurt, J., Hansen, H.O., Kronvang, B., Larsen, S.E., Rebsdorf, A. and Svendsen, L.M. (1994) *Vandmiljøplanens Overvågningsprogram 1993.* Ferske vandområder, vandløb og kilder. Faglig Rapport fra Danmarks Miljøundersøgelser 119. pp. 186.

Gustafson, A. (1982) Växtnäringsförluster från åkermark i Sverige. *Ekohydrologi,* **11**, 19-27.

Hammer, D.A. (1992) Designing constructed wetlands systems to treat agricultural nonpoint source pollution. *Ecological Engineering,* **1**, 49-82.

Hartikainen, H. (1979) Effect of cumulative fertilizer dressings on the phosphorus status of mineral soils. II. Comparison of two phosphorus testing methods. *Journal of the Agricultural Science in Finland,* **61**, 2, 61-66.

Hartikainen, H. (1991) Potential mobility of accumulated phosphorus in soil as estimated by the indices of Q/I plots and by extractant. *Soil Science,* **152**, 204-209.

Heathwaite, A.L., Burt, T.P. and Trudgill, S.T. (1990) Land-use controls on sediment production in a lowland catchment, South-West England. In: Boardman, J., Foster, I.D.L. and Dearing, J.A. (Eds.) *Soil erosion on agricultural land.* Proceedings of a workshop sponsored by the British Geomorphological Research Groups, Coventry, UK. pp. 69-86.

Hirvonen, A.T. (1994) Can multi-use ponds reduce the phosphorus transport to lakes in agricultural areas? *Verhandlungen der Internationale Vereinigung für theoretische und angewandte Limnologie,* **25**, 1322-1323.

Hirvonen, A., Helminen, H. and Salonen, V-P. (1996) Laskeutusallas pelto-ojien fosfori- ja kiintoainekuormituksen vähentäjänä – tutkimustulokset Köyliöstä. Abstract: Can a multi-use pond reduce the phosphorus and suspended solids transported to lakes in agricultural areas. *Vesitalous,* **3/96**, 2-5,46.

Hörberg, I. and Kylefors, L. (1991) Reduction of nutrients by water plants – Final report the Snogeröd project. (In Swedish with English abstract.) *Vatten,* **47**, 310-314.

Jenssen, P.D., Mæhlum, T. and Krogstad, T. (1993) Potential use of constructed wetlands for wastewater treatment in northern environments. *Wat. Sci. Tech.,* **28**(10), 149-157.

Jenssen, P.D., Mæhlum, T., Roseth, R., Braskerud, B., Syversen, N., Njs, A. and Krogstad, T. (1994) The potential of natural ecosystem self-purifying measures for controlling nutrient inputs. *Marine Pollution Bulletin,* **29**(6-12), 420-425.

Kronvang, B. and Svendsen, L.M. (1991) Phosphorus supply to the soil-freshwater environment in the Nordic countries. In: Svendsen, L.M. and Kronvang, B. (Eds.) *Phosphorus in the Nordic Countries – methods, bioavailability, effects and measures.* (In Danish with English summary, legends and headings.) Nordisk Ministerråd, København Nord 1991, 47, pp. 101-108.

Lee, D., Dillaha, T.A. and Sherrard, J.H. (1989) Modelling phosphorus transport in grass buffer strips. *Journal of Environmental Engineering,* **115**, 409-427.

Leonardson, L. (1994) *Wetlands as nitrogen sinks, Swedish and international experience.* (In Swedish with English summary.) SNV Rapport 4176.

Lindkvist, H. (1992) *Kväve-, fosfor- och partickelfångande egenskaper hos en halländsk damm under ett höstflöde.* (In Swedish with English abstract.) Swedish University of Agricultural Sciences, Division of Water Management. Seminarier och examensarbeten 16. Uppsala. pp. 27.

Lindkvist, H. and Håkansson, Å. (1993) *Kväve-, fosfor- och partickelfångande egenskaper hos en halländsk damm under ett vårflöde.* (In Swedish with English abstract.) Swedish University of Agricultural Sciences, Division of Water Management. Seminarier och examensarbeten 21. Uppsala. pp. 21.

Löfgren, S. and Olsson, H. (1990) *Tillförsel av kväve och fosfor till vattendrag i Sveriges inland* – Underlagsrapport till Hav -90, Aktionsprogram mot havsföroreningen. (In Swedish with English summary.) Naturvårdsverkets rapport 3692.

Logan, T.J. (1982) Mechanisms for release of sediment-bound phosphate to water and the effects of agricultural land management on fluvial transport of particulate and dissolved phosphate. *Hydrobiologia,* **92**, 519-530.

Lowrance, R.R. (1991) Effects of buffer systems on the movement of N and P from agriculture to streams. In: *NPO-forskning fra Miljstyrelsen,* Conference Contributions, International Conference on N, P and Organic Matter, Miljstyrelsen, Helsingr, Denmark. pp. 87-96.

Magette, W.L., Brinsfield, R.B., Palmer, R.E., Wood, J.D., Dillaha, T.A. and Reneau, R.B. (1987) *Vegetated filter strips for agricultural runoff treatment.* CBP/TRS 2/87, U.S. Environmental Protection Agency, Philadelphia, PA 19107. pp. 125.

Magette, W.L., Brinsfield, R.B., Palmer, R.E. and Wood, J.D. (1989) Nutrient and sediment removal by vegetated filter strips. *Transactions of the American Society of Agricultural Engineers,* **32**, 663-667.

Mander, Ü., Matt, O. and Nugin, U. (1991) Perspectives on vegetated shoals, ponds and ditches as extensive outdoor systems of wastewater treatment in Estonia. In: Etnier, C. and Guterstam, B. (Eds.) *Ecological Engineering for wastewater treatment.* Proceedings of the International Conference at Stensund Folk College, Sweden. pp. 271-282.

Mitsch, W.J. (1992) Landscape design and the role of created, restored and natural riparian wetlands in controlling nonpoint source pollution. *Ecological Engineering,* **1**, 27-47.

Osborne, L.L. and Kovacic, D.A. (1993) Riparian vegetated buffer strips in water-quality restoration and stream management. *Freshwater Biology,* **29**, 243-258.

Persson, P. (1990) Utilization of phosphorus in suspended particulate matter as tested by algal bioassays. *Verhandlungen der Internationale Vereinigung für theoretische und angewandte Limnologie,* **23**, 242-246.

Peterjohn, W.T. and Correll, D.L. (1984) Nutrient dynamics in an agricultural watershed: observations on the role of a riparian forest. *Ecology,* **65,** 1466-1475.

Pietiläinen, O.-P. and Rekolainen, S. (1991) Dissolved reactive and total phosphorus load from agricultural and forested basins to surface waters in Finland. *Aqua Fennica,* **21**, 127-136.

Pommel, B. and Dorioz, J.M. (1995) *Movement of phosphorus from agricultural soil to water.* Paper presented at an international Workshop "Phosphorus loss to water from agriculture", (September 27th to 29th), Teagasc, Johnstown Castle, Wexford, Ireland. (Dr. B. Pommel, Laboratoire d'Agronomie, INRA, Centre de Grignon, 78850 Thivernal-Grignon, France.) pp. 14.

Priha, M. 1994. Bioavailability of pulp and paper mill effluent phosphorus. Water Science and Technology 29:93-103.

Rekolainen, S., Pitkänen, H., Bleeker, A. and Felix, S. (1995) Nitrogen and phosphorus fluxes from Finnish agricultural areas to the Baltic Sea. *Nordic Hydrology,* **26**, 55-72.

Richardson, C. (1985) Mechanisms controlling phosphate retention capacity in freshwater wetlands. *Science,* **228**, 1424-1427.

Schwer, C.B. and Clausen, J.C. (1989) Vegetative Filter Treatment of Dairy Mikhouse Wastewater. *Journal of Environmental Quality,* **18**, 446-451.

Shannon, E.E. (1983) *Investigation of bioavailable phosphorus (BAP) in municipal wastewaters and the effects of wastewater treatment on BAP.* Final report submitted to Environment Canada by Canviro Consultants, Ltd. Kitchener, ONT.

Sharpley, A.N. (1985) Depth of surface soil-runoff interaction as affected by rainfall, soil slope and management. *Soil Sci. Soc. Am. J.,* **49**, 1010-1015.

Sharpley, A.N. and Syers, J.K. (1979) Phosphorus inputs into a stream draining an agricultural watershed. II: Amounts contributed and relative significance of runoff types. *Water Air Soil Pollut.,* **11**, 417-428.

Sharpley, A.N., Smith, S.J., Jones, O.R., Berg, W.A. and Coleman, G.A. (1992) The transport of bioavailable phosphorus in agricultural runoff. *Journal of Environmental Quality,* **21**, 30-55.

Svendsen, L.M. (1992) *Dynamics of phosphorus, organic nitrogen and organic matter in watercourses – methods, retention, transport and models.* Ph.D. thesis. Department of Freshwater Ecology, National Environmental Research Institute, Denmark. 260 p.

Syversen, N. (1994) Effect of vegetative filter strips on minimizing agricultural runoff in southern Norway. In: Persson, R. (Ed.) *Agrohydrology and nutrient balances.* Proceedings of NJF-seminar 247, Division of Agricultural Hydrotechnics, Swedish University of Agricultural Sciences, Uppsala, Sweden. Communications 94:5 pp. 70-74.

Taponen, T. (1995) *Laskeutusaltaat maatalouden vesiensuojelussa.* (Sedimentation basins in water protection control of agriculture.) Uudenmaan ympäristökeskus, Julkaisuja 3, Helsinki, pp. 56.

Timmons, D.R., Burwell, R.E. and Holt, R.F. (1973) Nitrogen and phosphorus losses in surface runoff from agricultural land as influenced by placement of broadcast fertilizer. *Water Resour. Res.,* **9**, 658-667.

Turtola, E. (1994) Phosphorus leaching from set-aside on clay soil. In: Jensen, H.E., Shjønning, P., Mikkelsen, S.A. and Madsen, K.B. (Eds.) *Soil Tillage for Crop Production and Protection of the Environment.* Proceedings of 13th International Conference, International Soil Tillage Organization, Aalborg, Denmark.Volume I: 299-303.

Turtola, E. and Jaakkola, A. (1995) Loss of phosphorus by surface runoff and leaching from a heavy clay soil under barley and grass ley in Finland. *Acta Agriculturae Scandinavica B, Soil and Plant Science,* **45**, 159-165.

Turtola, E. and Paajanen, A. (1995) Influence of improved subsurface drainage on phosphorus losses and nitrogen leaching from a heavy clay soil. *Agricultural Water Management,* **28**, 295-310.

Uhlen, G. (1978) *Nutrient leaching and surface runoff in field lysimeters on a cultivated soil.* I. Runoff measurements, water composition and nutrient balances. Scientific Reports of the Agricultural University of Norway 57.

Uhlen, G. (1988) Surface runoff losses of phosphorus and other nutrient elements from fertilized grassland. *Norwegian Journal of Agricultural Sciences,* **3**, 47-55.

Uusi-Kämppä, J. and Yläranta, T. (1996) Effect of buffer strips on controlling soil erosion and nutrient losses in southern Finland. In: Mulamoottil, G., Warner, B.G. and McBean, E.A. (Eds.) *Wetlands: Environmental Gradients, Boundaries and Buffers.* New York, CRC, Lewis Publishers. pp. 221-235.

Vought, L.B.-M., Dahl, J., Pedersen, C.L. and Lacoursière, J.O. (1994) Nutrient retention in riparian ecotones. *Ambio*, **23**, 342-348.

Walton, C.P. and Lee, G.F. (1972) A biological evaluation of the molybdenum blue method for orthophosphate analysis. *Verhandlungen der Internationale Vereinigung für theoretische und angewandte Limnologie,* **18**,6, 676-684.

Vanek, V. (1991) Riparian zone as a source of phosphorus for a groundwater-dominated lake. *Water Res.,* **25**, 409-418.

Williams, R.D., Nicks, A.D. and Krider, J.N. (1990) *Evaluation of vegetative filter strips as a management practice for non-point source pollution.* Environmental Engineering Proceedings. p. 799-806.

Yli-Halla, M., Hartikainen, H., Ekholm, P., Turtola, E., Puustinen, M. and Kallio, K. (1995) Assessment of soluble phosphorus load in surface runoff by soil analyses. *Agric. Ecos. Environ.*, **56**, 53-62.

Yli-Halla, M. and Hartikainen, H. (1996) Release of soil phosphorus during runoff as affected by ionic strength and temperature. *Agricultural Food Science in Finland,* **5**, 193-202.

Young, T.C., DePinto, J.V., Flint, S.E., Switzenbaum, M.S. and Edzwald, J.K. (1982) Algal availability of phosphorus in municipal wastewater. *Journal of the Water Pollution Control Federation,* **54**, 1505-1516.

6 Nitrogen dynamics and buffer zones

J.W. Gilliam, J.E. Parsons and R.L. Mikkelsen

North Carolina State University, Department of Soil Science, Box 7619, Raleigh 27695-7619, USA

Abstract

Riparian buffers have been proven to be very effective in the removal of sediment associated nitrogen from surface runoff and nitrate from subsurface flows. In both surface and subsurface flows, hydrologic characteristics are the key to determining how effective the buffer will be in nitrogen removal. Even though buffers are extremely important to minimise entry of non-point source nitrogen into surface waters and removals of 90% are common, they do not work well in some hydrologic conditions.

INTRODUCTION

Permanent vegetation between agricultural fields, grazed land or forests and surface waters has long been known to be effective in reducing movement of sediment and nutrients to surface waters. Although this information was much more qualitative than quantitative, vegetative filter strips have long been accepted as a Best Management Practice for conservation of soil and water resources. Historically, justification for the practice was based on limiting movement of sediment in surface runoff water. This changed in the late 1970's when the importance of streamside vegetation on stream environment and health was recognised (Karr and Schlosser, 1978; Johnson and McCormick, 1978). Water quality improvement by riparian buffers was recognised in a general manner by a few authors (Kitchens *et al.*, 1975; Kuenzler *et al.*, 1977). However, documentation of water quality improvement was still largely limited to surface processes. In the early 1980's, three independent reports indicated that water quality improvements by riparian buffers also improved subsurface water quality (Lowrance *et al.*, 1983; Peterjohn and Correll, 1984; Jacobs and Gilliam, 1985). These authors showed that nitrate concentrations in shallow groundwater flowing through riparian buffers toward streams in the Atlantic Coastal Plain of the US were reduced by as much as 90%.

The ever increasing concern with water quality and general recognition of the water quality benefits of riparian buffers has resulted in a large increase in the amount of research on this topic in the past ten years (Gregory *et al.*, 1991). Even with limited quantitative information, there has also been a widespread acceptance by the general public and regulators of the water quality benefits of riparian buffers. Many regulatory bodies are currently encouraging increased use of riparian buffers through voluntary or regulatory procedures. Although the focus of this paper is on nitrogen, overall water quality benefits of riparian buffers must be considered because of the questions currently being asked by the public as to the best use of riparian buffers.

There have been several recent reviews on riparian buffers which focus largely on nitrogen as related to water quality (Gilliam, 1994; Lowrance *et al.*, 1995; Hill, 1996) so this paper will not attempt a complete review of the literature. The excellent review by Hill (1996) gives a detailed description of the current status of knowledge on the removal of nitrate in riparian buffers. Thus this paper will not attempt to duplicate that review but will review only that information necessary to explain what additional information and research is needed to answer the two major questions being asked throughout the world. These questions are "how wide do riparian areas need to be?" and "what plants are needed to achieve a certain level of treatment of both surface and subsurface flows?". To answer these questions accurately, a greater understanding of the interaction of the many factors controlling nitrogen removal by riparian buffers present in various landscapes is required.

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0

NITROGEN REMOVAL FROM WATER BY RIPARIAN BUFFERS

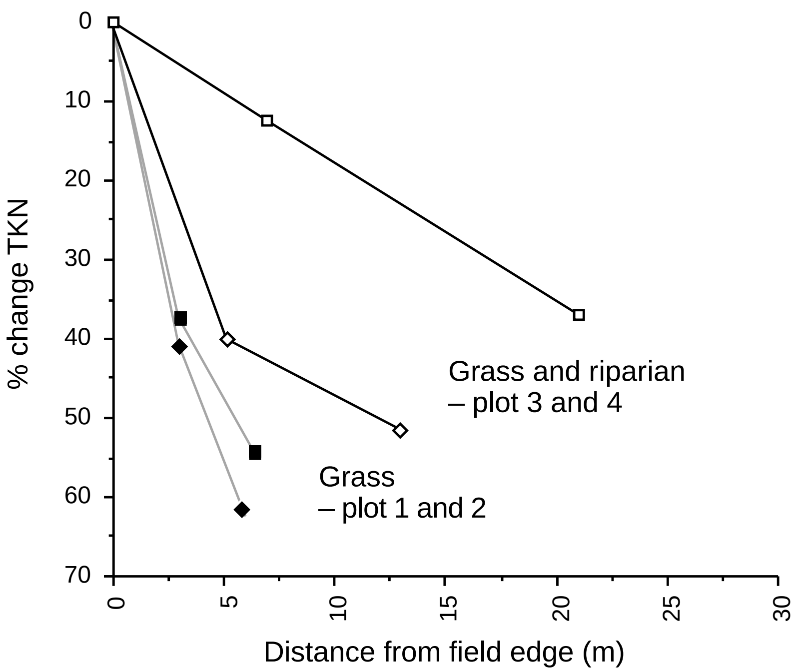
Most recent research papers and reviews on N removal from water in riparian buffers have focused on removal of nitrate from subsurface flows. However, nitrogen entering streams through surface runoff is also important and riparian buffers have an important role in removal of nitrogen from surface waters.

Surface runoff water

Unless heavy rain immediately follows a surface application of inorganic fertiliser to an agricultural field, most nitrogen in surface runoff is present as organic nitrogen associated with suspended solids. There is considerable information available about the TKN (organic-N plus NH_4-N) removal efficiency of grass buffers when runoff water from land receiving either animal waste (Bingham *et al.*, 1980) or inorganic fertilisers passes through them (Magette *et al.*, 1989; Daniels and Gilliam, 1996). Some of the researchers who first measured nitrate removal from subsurface water in wooded riparian buffers also measured nitrogen losses from surface water (Correll, 1983; Lowrance *et al.*, 1985; Chescheir *et al.*, 1991).

All of the above have measured very significant removals of N from surface waters flowing through buffers. As expected, removal of sediment-bound N is generally higher than removal of inorganic N in solution (Dillaha *et al.*, 1989). Also, N removals are quite variable between and within buffers as shown in Figure 1. Factors which influence sediment removal in buffers is covered in much more detail by Dillaha and Inamdar in this volume. Thus, we will not comment further on this except to point out that published (Parsons *et al.*, 1994) and unpublished data we have collected indicate that grass buffers are more effective in both sediment and TKN removals from surface water than forested buffers (when the buffers are of similar widths).

Figure 1. Loss of TKN as surface runoff water flows through grass and grass plus mixed vegetation (from Daniels and Gilliam, 1996).



Subsurface drainage water

The most studied nutrient removal process in riparian buffers during the past decade has been nitrate loss from subsurface flows. As pointed out by Hill (1996), there are several reasons for the tremendous interest in this process. These include: (a) the major pathway for nitrate movement to surface waters from many humid uplands is through diffuse subsurface flow which may or may not pass through a riparian buffer, (b) there is general agreement among many researchers that presence of riparian buffers is the most important factor controlling entry of non-point source nitrate in surface water (Lowrance *et al.*, 1995) and (c) there was essentially no scientific information about the process before the early 1980's. However, there are still many unanswered important questions making it difficult to predict nitrate losses under many landscape situations. There is agreement that the two major processes responsible for removal of nitrate in riparian buffers are plant uptake and denitrification but there is no agreement of the relative importance of these two processes in many landscapes. There is no question that both are important processes whose relative importance varies with landscape conditions. All who have looked at a variety of riparian buffers throughout the world agree that they are complex environments which are heterogeneous in both the vertical and horizontal direction in all of the important characteristics controlling nitrate loss.

Many early studies on nitrate removal in riparian buffers measured a decrease in nitrate concentration in the groundwater with distance from a field edge into the buffer without directly measuring the process(es) responsible for the decrease. Denitrification was assumed by some because reducing conditions were measured in the saturated sediments near the stream and there was a marked decease in the NO_3-N/Cl ratio (Jacobs and Gilliam, 1985). Also, the first few studies were all conducted in the Atlantic Coastal Plain of the US in areas where a relatively shallow aquatard existed which was assumed to prevent both (i) movement of significant amounts of water upward to dilute nitrate concentration in the laterally flowing groundwater and (ii) movement of nitrate to deeper groundwater. Later studies by Schipper *et al.,* (1993, 1994) in New Zealand, Haycock and Burt (1993) in Great Britain and Pinay *et al.,* (1993) in France very clearly showed that denitrification was the primary mechanism responsible for nitrate disappearance in their riparian buffers.

However there are problems with assuming that denitrification will remove most of the nitrate wherever groundwater flows below a riparian buffer. In almost every study cited thus far, the nitrate concentration decreased dramatically in the first few (<10) metres of the buffer. In these situations, it would seem that uptake could not be responsible for all of the nitrate disappearance occurring within this short distance. However, the water table at the upper edge of the buffer in our studies in North Carolina and in Lowrance's in the Coastal Plain of Georgia was commonly more than a metre below the surface of the soil. Groffman *et al.*, (1991) and Lowrance (1992) have clearly shown that C availability for denitrification in the soil below the A horizon in some of the riparian soils where they measured apparent nitrate removals was limiting (Table 1). These data do not indicate that denitrification would not occur at the depths where most water is flowing through the edge of the buffer, but losses are apparently occurring where measured denitrification potentials are low. We have collected similar unpublished data in North Carolina showing that nitrate is apparently being lost even when denitrification rates were low. Groffman (personal communication) has hypothesised that much of the denitrification occurring riparian buffers with apparent low denitrification potential may be occurring in localised micro sites as has been measured in field soils by Parkin (1987).

Table 1. Changes in groundwater nitrate concentration, depth to water table, and denitrification potential in the first 10 m of a riparian buffer in Georgia (from Lowrance, 1992).

NO_3-N Conc. (mg/L)		*Water Table Depth (m)*			*Relative Denitrification Potential at Buffer Edge*		
Buffer Edge	*In 10 m*	*Max.*	*Min.*	*Mean*	*Soil Depth (cm)* 5	30	90
14	2	2	0.6	1.6	1	0.07	0.01

Many authors have noted that knowledge of hydrology is essential to understanding nitrate removal rates in riparian buffers (Correll and Weller, 1989; Hill, 1990; Gilliam, 1994) although few have made detailed studies of the local hydrology. Hill (1996) pointed out many deficiencies in our current knowledge in this area. Two recent studies show how important hydrologic information can be in interpreting results of riparian buffer studies. Altman and Parizek (1995) measured large changes in nitrate concentration in the groundwater near the discharge area from a farm in the Valley and Ridge and the Appalachian Plateau of the US. They determined that a small part of the decreased concentration may have been a result of denitrification, but the change was mostly due to dilution by groundwater from another area. These authors were careful to point out the hydrologic differences between their study area and other areas studied in the Atlantic Coastal Plain to help explain why their results were different. However, many reviewers have questioned results of any study where denitrification was not directly measured and/or age of groundwater determined. Thus it is important that researchers working on N loss in riparian buffers obtain sufficient information to show exactly what the N dynamics are.

A very interesting technique for dating groundwater by determining chlorofluorocarbons concentration in the water was recently developed by US Geological Survey (Dunkle *et al.,* 1993; Plummer *et al.,* 1993). Puckett *et al.,* (1995) used this technique in a recent study in a Minnesota riparian wetland to show that water in the riparian area containing very low nitrate concentrations was much older than the groundwater at the edge of the field which contained much higher levels of nitrate. Because the groundwater in the riparian wetland was so much older than groundwater below nearby fields, nitrate concentration differences probably were not a result of nitrate removal in the riparian buffer. The technique of dating groundwater and its utilisation in riparian buffer studies was presented by Puckett as a poster at this conference and should be applied in other riparian areas to better understand the dynamics controlling movement of nitrate to surface waters.

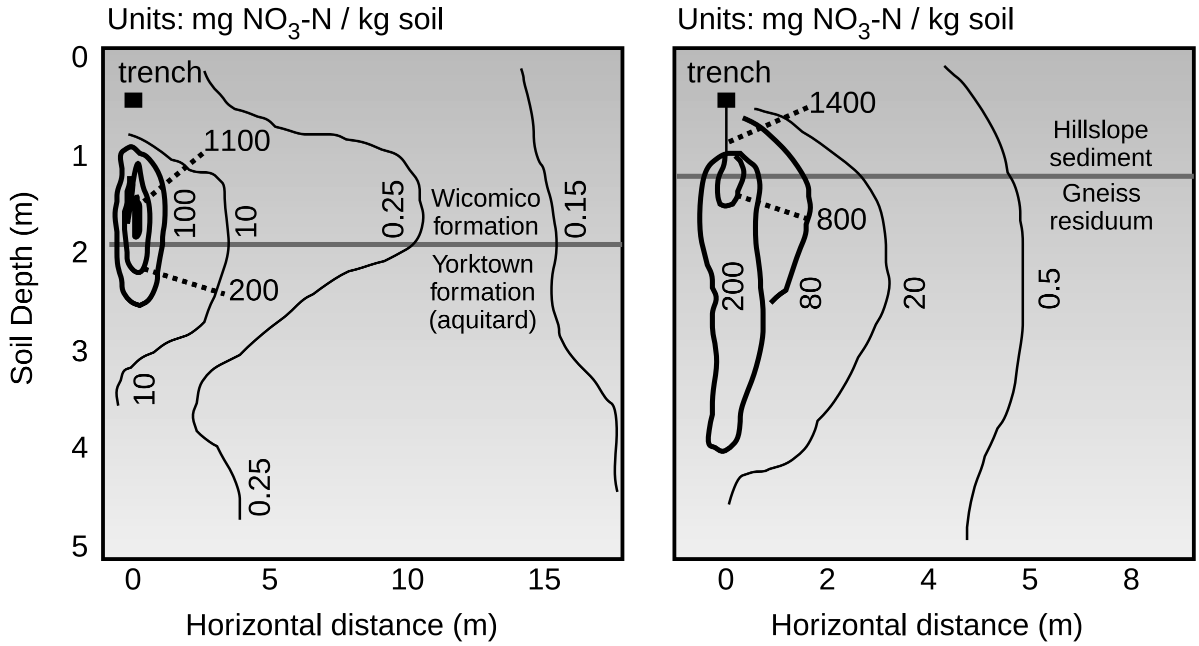
Scientists working with nitrate loss in groundwater below riparian buffers have discussed the possibility of the water passing below a riparian buffer at depths where there would be no or minimal influence of the buffer. Data to confirm this possibility were recently obtained in the Delmarva Peninsula by Phillips *et al.,* (1993) and Correll *et al.,* (1994). Their data indicate that the lack of a shallow aquitard may greatly reduce nitrate removal from groundwater in riparian areas. Thus one cannot assume that a riparian buffer will necessarily remove nitrate from groundwater flowing below it.

Another aspect of hydrology influencing nitrate removal in riparian areas is residence time of the water in the buffer. This was demonstrated very well by Haycock and Pinay (1993) who showed that nitrate in groundwater entering a riparian floodplain at a fast flow rate moved further into the floodplain than nitrate at a similar concentration in groundwater moving slower.

Because of the low denitrification potentials observed in subsoils where nitrate concentrations were decreasing in riparian buffer groundwater, we tried to estimate residence time of the nitrate in the riparian zone (Xu, 1992). Nitrate and chloride salts were placed in trenches at the top of the B horizon upslope of riparian buffers at the Piedmont site (soil: Clayey, kaolinitic, thermic Typic Hapludult) and at the Coastal Plain site (soil: Fine-loamy, siliceous, thermic Typic Paleudult) mentioned earlier. The vertical and horizontal movement of the nitrate in the riparian areas after 530 days and 1147 mm of rain at the Piedmont site and 558 days and 1366 mm of rain at the Coastal Plain site are shown in Figure 2. The movement was slow at both sites in both the vertical and horizontal directions. As expected, movement was faster in the sandy Coastal Plain soil but residence time in the riparian buffer was relatively long. Under these hydrologic conditions, a slow rate of denitrification or plant uptake could remove a significant amount of nitrate before it left the area in subsurface drainage water.

Some have questioned how long riparian buffers can be effective in removing nitrate from ground water. Where most riparian buffers are currently effective, they are receiving continuing inputs of organic carbon from surface vegetation. Thus, they are likely to continue their effectiveness for a long period of time. However, Groffman *et al.,* (1992) suggested that long-term buffering at a site they studied may be limiting. Thus, a definitive answer to this question is currently lacking and may be site specific.

Figure 2. Nitrate distribution in a North Carolina Coastal Plain buffer 558 days after application (left side) and in a Piedmont buffer after 530 days (right side) (from Xu, 1992).



A very important question being asked by those encouraging increased protection of surface waters by riparian buffers concerns the relative removal efficiency of trees versus grass. Osborne and Kovaic (1993) in Illinois and Haycock and Pinay (1993) in England compared nitrate removal rates in buffers below trees and grass and both researchers reported somewhat higher removal in buffers that included trees. There has also been a general consensus among several scientists that trees are the preferred vegetation in a riparian buffer for nitrate removal because of a potentially deeper rooting zone (Lowrance *et al.*, 1995). Presumably, the deeper roots increases both plant uptake and C supply required for denitrification (Haycock and Pinay, 1993). This certainly may be true, but other scientists have pointed out that grasslands tend to have more organic matter deeper in the soil profile than do forests. Also, there are some deep-rooted grasses which might do better than those previously tested. There is very strong opposition by land owners in many areas to planting trees in riparian areas whereas a permanent grass buffer is much more acceptable.

Because environmental problems are not limited to water quality, concern is frequently expressed that increased denitrification in riparian areas may be trading a water quality problem for an atmospheric problem because of increased N_2O evolution. However, Weller *et al.*, (1994) measured lower N_2O evolution in their riparian buffers (0.35 kg N $ha^{-1}yr^{-1}$) in Maryland than was emitted from adjacent cropland. Buffington (1994) measured much larger amounts of N_2O evolution from our Piedmont and Coastal Plain sites in North Carolina than those measured in Maryland. However, the N_2O evolution rates were still similar to those measured from cropland in North Carolina (Spooner, 1980). Surprisingly, the N_2O concentration in the groundwater entering the riparian areas in Buffington's study contained about 200-500 μgL^{-1} N_2O-N at the Coastal Plain site and 40-100 μgL^{-1} N_2O-N at the Piedmont site. Concentrations decreased with distance into the riparian buffers at both locations. The data from both Maryland and North Carolina seem to indicate that increased use of riparian buffers to remove NO_3-N from groundwater is not likely to contribute much, if any, to the current problems caused by N_2O.

DISCUSSION

The authors believe that riparian buffers are the most important factor controlling entry of N, particularly NO_3-N, from agricultural land into surface water in humid regions. However, there is tremendous variation in their effectiveness even within a single buffer and certainly between physiographic regions. The public and regulatory agencies have accepted the scientific conclusion as to the value for protecting water quality. In many ways, the demand for information and utilisation of the practice is advancing faster than the science.

Welsch (1991) presented an idealised riparian buffer system for the protection of water quality. This general concept was also adopted by the group of scientists who evaluated the potential for use of riparian forest buffer systems to protect water quality in the Chesapeake Bay of the US (Lowrance *et al.*, 1995). The width between cropland and stream for the idealised system was 29 m. There is tremendous resistance by land owners where no vegetated buffer currently exists, to installing a 29 m buffer. We believe that a 29 m buffer is a desirable goal for wildlife, stream health, etc., but frequently many of the water quality benefits can be achieved with narrower buffers. For example, many studies have shown dramatic decreases in TKN in surface runoff water with a 10 m buffer and the same reductions have been seen for NO_3-N concentrations in subsurface flows. Current information allows one to design ideal riparian buffers, but we cannot predict results which would be obtained by less than ideal widths which may be greatly over-designed for many places. The same uncertainties may be true with regard to use of trees or grass. If a combination of trees and grass is best (Welsch, 1991), what are relative water quality benefits of varying widths of only trees or only grass?

Another question frequently asked about riparian buffers is in regard to channellised streams or large ditches. Frequently the water table is 1-3 m below the ground surface near the ditch bank so there might be little interaction between the water table and buffer vegetation. We have initiated research to see if we can use a combination of controlled drainage (structure in ditch or stream to regulate flow) and permanent vegetation on bank to enhance NO_3-N loss in these situations. Again, we have the unanswered question related to width of the vegetated strip. We are also developing simulation models to address the hydrology-nitrogen interaction in these areas. Our modelling effort will enable an examination of the relationship between riparian buffer width and nitrogen reduction in various landscapes. In addition we will be testing and evaluating other modelling approaches including the Riparian Ecosystem Management Model (REMM) which is currently being developed in Georgia (Bosch *et al.*, 1996).

REFERENCES

Altman, S.J. and Parizek, R.R. (1995) Dilution of non-point-source nitrate in groundwater. *J. Environ. Quality,* **24**, 707-718.

Bingham, S.C., Westerman, P.W. and Overcash, M.R. (1980) Effect of grass buffer zone length in reducing the pollution from land application areas. *Trans. Amer. Soc. Agric. Eng.,* **23**, 330-336.

Bosch, D.D., Williams, R.G., Sheridan, J.M. and Lowrance, R.R. (1996) *Riparian ecosystem management model: Sediment transport.* Paper no. 962021. Am. Soc. Ag. Eng. Int. Summer Meeting, Phoenix, AR. July 14-18.

Buffington, D.E. (1994) *Nitrous Oxide Dynamics and Denitrification in Four North Carolina Riparian Systems.* Unpublished M.S. Thesis. North Carolina State University, Raleigh, NC.

Chescheir, G.M., Gilliam, J.W., Skaggs, R.W. and Broadhead, R.G. (1991) Nutrient and sediment removal in forested wetlands receiving pumped agricultural drainage water. *Wetlands,* **11**, 87-103.

Correll, D.L. (1983) N and P in soils and runoff of three Coastal Plain land uses. In: Lowrance, R. *et al.* (Eds.) *Nutrient cycling in Agricultural Ecosystems.* Univ. of Georgia Coll. of Agric. Special Publ. No. 23. Athens, GA. p. 207-224.

Correll, D.L., Jordan, T.E. and Weller, D.E. (1994) Failure of agricultural riparian buffers to protect surface waters from groundwater contamination. In: Gibert, J. (Ed.) *Groundwater-Surface Water Ecotones.* Lyon, France. July, 1993 UNESCO/MAB/IHP., Cambridge Univ. Press.

Correll, D.L. and Weller, D.E. (1989) Factors limiting processes in freshwater wetlands: An agricultural primary stream riparian forest. In: Sharitz, R.R. and Gibbons, J.W. (Eds.) *Freshwater Wetlands and Wildlife.* Oak Ridge, TN. pp. 9-23.

Daniels, R.B. and Gilliam, J.W. (1996) Sediment and chemical load reduction by grass and riparian filters. *Soil. Sci. Soc. Am. J.,* **60**, 246-251.

Dillaha, T.A., Reneau, R.M., Mostaghimi, S. and Lee, D. (1989) Vegetative filter strips for agricultural non-point source pollution control. *Trans. Amer. Soc. Agric. Eng.,* **32**, 513-519.

Dunkle, S.A., Plummer, L.N., Budsenberg, E., Phillips, P.J., Denver, J.M., Hamilton, P.A., Michel, R.L. and Coplen, T.B. (1993) Chlorofluorocarbons (CCl_3F and CCl_2F_2) as dating tools and hydrologic tracers in shallow groundwater of the Delmarva Peninsula, Atlantic Coastal Plain, United States. *Water Res. Res.*, **29**, 3837-3860.
Gilliam, J.W. (1994) Riparian wetlands and water quality. *J. Environ. Qual.*, **28**, 896-900.
Gregory, S.V., Swanson, F.J., McKee, W.A. and Cummins, K.W. (1991) An ecosystem perspective of riparian zones. *Biosci.*, **41**, 540- 551.
Groffman, P.M., Axelrod, E.A., Lemunyon, J.L., and Sullivan, W.M. (1991) Denitrification in grass and forest vegetated filter strips. *J. Environ. Qual.*, **20**, 671-674.
Groffman, P.M., Gold, A.J. and Simmons, R.C. (1992) Nitrate dynamics in riparian forests: Microbial studies. *J. Environ. Qual.*, **21**(4), 666-671.
Haycock, N.E. and Pinay, G. (1993) Groundwater nitrate dynamics in grass and poplar vegetated riparian buffer strips during the winter. *J. Environ. Qual.*, **22**, 273-278.
Haycock, N.E. and Burt, T.P. (1993) Role of floodplain sediments in reducing the nitrate concentration of subsurface run-off: A case study in the Cotswolds, *UK. Hydrol. Processes,* **7**, 287- 295.
Hill, A.R. (1990) Ground water flow paths in relation to nitrogen chemistry in the near-stream zone. *Hydrobiologia,* **206**, 39-52.
Hill, A.R. (1996) Nitrate removal in stream riparian zones. *J. Environ. Qual.*, **25**, 743-755.
Jacobs, T.J. and Gilliam, J.W. (1985) Riparian losses of nitrate from agricultural drainage waters. *J. Environ. Qual.*, **14**, 472-478.
Johnson, R.R. and McCormick, J.F. (Eds.) (1978) *Strategies for protection and management of floodplain wetlands and other riparian ecosystems.* Gen. Tech. Rep. WO-12, Forest Serv., U.S. Dept. Agric., Wash. DC. 410 pp.
Karr, J.R. and Schlosser, I.J. (1978) Water resources and the land-water interface. *Science,* **201**, 229-234.
Kitchens Jr., W.M., Dean, J.M., Stevenson, L.H. and Cooper, J.H. (1975) The Santee Swamp as a nutrient sink. In: Howell, F.G., Gentry, J.B. and Smith, M.H. (Eds.) *Mineral Cycling in Southeastern Ecosystems.* Aiken, GA: Savannah River Ecology Laboratory. pp 349-366.
Kuenzler, E.J., Mulholland, P.J., Ruley, L.A. and Sniffen, R.P. (1977) *Water quality in North Carolina Coastal Plain streams and effects of channelisation.* Water Resources Research Institute, UNC. Rpt. No. 127. Raleigh, NC. 160 pp.
Lowrance, R.R. (1992) Groundwater nitrate denitrification in a coastal plain riparian forest. *J. Environ. Qual.*, **21**, 401- 405.
Lowrance, R., Altier, L.S., Newbold, J.D., Schnabel, R.R., Groffman, P.M., Denver, J.M., Correll, D.L., Gilliam, J.W., Robinson, J.L., Brinsfield, R.B., Staver, K.W., Lucus, K. and Todd, A.H. (1995) *Water Quality Functions of Riparian Forest Buffer Systems in the Chesapeake Bay Watershed.* U.S.E.P.A. Chesapeake Bay Program. Technology Transfer Report. Annapolis, MD.
Lowrance, R., Leonard, R.A., Asmussen, L.E., and Todd, R.L. (1985) Nutrient budgets for agricultural watersheds in the southeastern coastal plains. *Ecology,* **66**, 287-296.
Lowrance, R.R., Todd, R.L. and Asmussen, L.E. (1983) Waterborne nutrient budgets for the riparian zone of an agricultural watershed. *Agric. Ecocsys. Environ.*, **10**, 371-384.
Magette, W.L., Brinsfield, R.B., Palmer, R.E. and Wood, J.D. (1989) Nutrient and sediment removal by vegetated filter strips. *Trans. Amer. Soc. Agric. Eng.*, **32**, 663-667.
Osborne, L.L. and Kovacic, D.A. (1993) Riparian vegetated buffer strips in water-quality restoration and stream management. *Freshwater Biol.*, **29**, 243-258.
Parkin, T.B. (1987) Soil microsites as a source of denitrification variability. *Soil Sci. Soc. Am. J.*, **51**, 1196-1199.
Parsons, J.E., Daniels, R.B., Gilliam, J.W. and Dillaha, T.A. (1994) *Reduction in sediment and chemical load from agricultural field runoff by vegetative filter strips.* Water Resources Research Institute, UNC Rpt No. 286, Raleigh, NC. 75pp.
Peterjohn, W.T. and Correll, D.T. (1984) Nutrient dynamics in an agricultural watershed: Observations on the role of a riparian forest. *Ecology.*, **65**, 1466-1475.
Phillips, P.J., Denver, J.M., Shedlock, R.J. and Hamilton, P.A. (1993) Effect of forested wetlands on nitrate concentrations in ground water and surface water on the Delmarva Peninsula. *Wetlands,* **13**, 75-83.
Pinay, G., Roques, L. and Fabre, A. (1993) Spatial and temporal patterns of denitrification in a riparian forest. *J. Appl. Ecol.*, **30**, 581-591.
Plummer, L.N., Michel, R.L., Thurman, E.M. and Glynn, P.D. (1993) Environmental tracers for age dating young ground water. In: Alley, F.M. (Ed.) *Regional Ground-Water Quality.* Van Nostrand Reinhold, NY. pp 255-294.
Puckett, L.J., Tornes, L.H., Cowdery, T.K., Brigham, M.E. and Stoner, J.D. (1995) *The role of wetlands in reducing nitrogen loadings to the Otter Tail River in Minnesota.* Amer. Geophy. Union. Fall Ann. Mtg. San Francisco, CA. Dec. 11-15.
Schipper, L.A., Cooper, A.B., Harfoot, C.G., and Dyck, W.J. (1993) Regulators of denitrification in an organic riparian soil. *Soil Biol. Biochem.*, **25**, 925-933.
Schipper, L.A., Cooper, A.B., Harfoot, C.G. and Dyck, W.J. (1994) An inverse relationship between nitrate and ammonium in an organic riparian soil. *Soil Biol. Biochem.*, **26**, 799-800.
Spooner, J. (1980) *Nitrous Oxide Evolution from Typic Umbraquult and Terric Medisaprist Soils.* Unpublished M.S. Thesis. North Carolina State University. Raleigh, NC.

Welsch, D.J. (1991) *Riparian Forrest Buffers – Function and design for protection and enhancement of water resources.* USDA Forest Service Pub. NA-PR-07-91. 20p.

Weller, D.E, Correll, D.L. and Jordan, T.E. (1994) Denitrification in riparian forests receiving agricultural discharges. In: Mitsch, W.J. (Ed.). *Global Wetlands: Old World and New.* pp. 117-131.

Xu, L. (1992) *Nitrate Movement and Removal in Riparian Buffer Areas.* Unpublished M.S. Thesis. North Carolina State University. Raleigh, NC.

7 Pesticide contamination of surface waters – the potential role of buffer zones

G.L. Harris[1] and A. Forster[2]

[1]*ADAS Land Research Centre, Meden Vale, Mansfield, Nottinghamshire NG20 9PF, UK.*
[2]*Huntingdon Life Sciences, PO Box 2, Huntingdon, Cambridgeshire PE18 6ES, UK.*

Abstract

The movement of pesticides to surface waters has become an area of concern across Europe and other countries where pesticide usage is a key part of crop management. Pesticide losses to surface waters can be rapid; as a consequence, remedial measures may have a more or less immediate effect in reducing contamination, and resulting environmental impact. One such measure attracting increasingly widespread interest is the use of buffers generally considered to be best located close to, or adjacent to, surface water courses. However, the mechanisms by which buffer zones can control pesticide loss are not well understood, neither is the optimum design and function of buffers always clear. This review paper considers the mechanisms and importance of pesticide transport to surface waters and assesses the evidence that indicates whether buffers can be effective in protecting both water quality and the environment. In particular, the paper examines research which addresses the appropriate design of buffers and assesses the potential long-term role for these landscape features.

INTRODUCTION

As agriculture has intensified, an increasingly wide range of pesticides has been found in surface waters draining agricultural catchments at concentrations above that considered appropriate for potable water and wildlife (White and Pinkstone, 1995). In the EU, Williams *et al.* (1991), Gillet (1991), Harris *et al.* (1993) and Traub-Eberhard *et al.* (1995) are among many who have reported studies showing appreciable pesticide losses. In the US, Asmussen *et al.* (1977), Neely and Baker (1989) and Bengston *et al.* (1990) all reported pesticides in surface runoff at concentrations considerably above $0.1\mu g/l$. Most workers have reported that peak pesticide concentrations were found soon after application and the initiation of surface runoff or drainflow, often with concentrations higher in the US than in the EU. Leonard (1988) in particular, suggested that these higher peak pesticide concentrations in the US were due to more intense storms than are found in Europe.

Pesticide movement has been reported both in solution and in the particulate phase. Flow paths for each can be different and the likelihood of a pesticide being absorbed to particulates will depend on its adsorption properties (Marshall *et al.*, 1996). As a consequence, measures introduced into the landscape to address pesticide movement must reflect these different flow paths.

The conversion of streamside margins and riparian areas to provide a buffer to pesticides reaching watercourses has been reported widely (Muscutt *et al.*, 1993; Norris, 1993). In particular, in the US, vegetated filter strips are an approved 'Best Management Practice' which is part-funded by the US Department of Agriculture. Also, in the US, widespread installation has occurred along all perennial streams (Dillaha, 1989). Other countries have also adopted similar approaches. For example, in some Scandinavian countries vegetated buffer zones are already widely used alongside lakes to control contamination (Keskitalo, 1990) whereas in New Zealand a policy of 'retirement' of riparian zones has been recommended to protect aquatic habitats (Smith, 1989).

This paper reviews the information available on the mechanisms of diffuse pesticide transport from agricultural soils and assesses the impact that buffers and a buffer policy could have on pesticide concentrations and the biodiversity of surface waters.

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0

MECHANISMS OF PESTICIDE MOVEMENT

Surface runoff

Surface runoff can occur when the surface soil becomes saturated or when rainfall intensity exceeds the infiltration capacity of the surface soil aggregates. Agricultural management can influence water movement. For example compaction of the soil surface by machinery or the use of permanent tramlines (Harris, 1995) can increase surface runoff.

Many pesticides are readily carried in surface runoff at concentrations that could affect both catchment water quality and stream biodiversity; this process applies particularly to pesticides only weakly adsorbed (Asmussen *et al.,* 1977; Wauchope and Decoursey, 1986; Leonard, 1990; Harris *et al.,* 1994; Brown *et al.*, 1995).

Sub-surface runoff

Sub-surface flow is widespread in many surface water catchments, particularly in clay-based soils without pipe drainage. Where permeability is particularly low, permanent pipe drains will be linked to a secondary drainage treatment to improve water movement (Harris, 1995). In such soils the presence of macropores is particularly important to the flow of water (Beven and Germann, 1982). In contrast in the US, and in more permeable EU soils, surface drainage, or drainage without secondary treatments, is more common and considerable research has focused in these regions on surface runoff (Leonard, 1990).

Pesticides are often applied in the autumn, just before the onset of drainage, or in the spring when soils are still relatively wet and hence rapid losses can occur. Once a pesticide is applied to the soil, the likelihood of movement to sub-surface drainage will depend on the properties of the chemical itself (especially mobility and degradation), soil structure and organic matter content, the period of time between application and drainflow, and the background soil moisture and temperature conditions (Jones *et al.,* 1995). However, some groups of pesticides are more likely to be lost than others, for example herbicides, such as isoproturon, which are readily mobile and remain in the soil for many months (Monke *et al.,* 1989; Williams *et al.,* 1991; Harris *et al.,* 1994). In contrast strongly adsorbed pesticides have only been found at very low concentrations and with low overall losses (Jones *et al.*, 1995).

Particulate transport

Wauchope (1978) has suggested that only pesticides with a solubility in excess of 10 ppm are likely to be lost primarily in the water phase. However, it is evident that many pesticides are found in particulate matter and that this may be an important transport route in some conditions. Buttle (1990) found 20-46% of the losses of the pesticide metachlor (a moderately soluble herbicide) were carried in sediments over the reporting period. House *et al.* (1992) and Worral *et al.* (1993) have also highlighted the importance of this mode of transport. House *et al.* (1992) found simazine and atrazine in water in three UK catchments, but lindane, DDT and other strongly adsorbed pesticides were also detected in bed and suspended sediments. Recent studies, for example those reported by Marshall *et al.* (1996), are investigating the source of the particulate matter involved and will be important in determining the most appropriate control measures for this route of pesticide transport.

Spray drift

A further source of contamination not always considered is that from spray drift, directly into the watercourse, which can pose a further risk to aquatic life. Numerous studies have shown that spray can drift over considerable distances (Harris *et al.,* 1992; Ganzelmeier, 1993; Lloyd and Bell, 1993), although for a typical arable crop the volume of the original spray application that travels in excess of 6 m may be no more than 1%.

POTENTIAL FOR BUFFER ZONES TO REDUCE PESTICIDE LOSSES

Surface runoff

Field margin buffer zones are widely reported to reduce diffuse contamination of watercourses, especially for nitrate movement and where water movement is relatively slow through the buffer area (Muscutt *et al.*, 1993; Norris, 1993; Patty *et al.*, 1995; Patty and Real, 1996). Other benefits have been reported for buffers, e.g. as wildlife refuges and to remove irregular field boundaries. However, research has also shown that uncropped areas can potentially reduce in-field crop yields (Speller *et al.*, 1992).

Plot studies in the US have suggested that grass buffers are effective sediment filters with retention of over 80% reported (Dillaha *et al.*, 1987; Parsons *et al.*, 1990). However, Niebling and Alberts (1979) found that buffers were less effective in the clay particle range, especially when only narrow strips were used. Fenessey (1993) suggested 40-80 m wide strips were needed to reduce surface runoff and contamination in poorly drained soils and 15-60 m wide strips for better drained soils. However, this does not take into account the problem of point source input, especially the importance of sub-surface drainage in clay soils (Harris *et al.*, 1994). US grant-aided filter strips are between 20 and 30 m wide, whilst German work generally recommends a buffer zone of 5-10 m on well-drained soils for the effective interception of surface runoff, and a width of 15-20 m for poorly drained soils.

Research into the effect of buffers in controlling pesticide movement is less well documented. Experiments in the US by Mickleson and Baker (1993) suggested that 4.6 m grass buffers could remove 72% of sediment (as long as the slope was not too steep to cause erosion near to the watercourse) and by 75% for 9.1 m buffers. The 4.6 m and 9.1 m buffers reduced losses of atrazine from a 4.6% slope by 32 and 55% respectively. Tillage on the upslope area did not affect the results. Baker *et al.* (1995) suggest that even 2 m wide buffer strips increased water infiltration and could therefore be effective in reducing pesticide movement. Jones (1993) reported similar German studies using 5 m buffers on a 13% slope. He found that although runoff was little affected by a bare soil buffer, herbicide losses were reduced by 40%. In contrast a grass buffer virtually eliminated soil erosion, reduced water movement and substantially cut pesticide movement to surface waters. He concluded that although more research was needed to quantify the size of buffers, they could be effective in reducing pesticide losses to surface waters.

More recent studies by Gril *et al.* (this volume) found that buffers were effective at removing sediment and a range of pesticides studied. Using a range of buffer sizes from 6 to 20 m, considerable reductions were achieved. For example, concentrations of isoproturon and diflufenican were reduced by at least 57% and 68% with 5.7 and 11.1 m buffers, respectively.

Sub-surface drainage

There is little reported evidence on the benefits of buffers in reducing pesticide movement in sub-surface drainage waters. This is because movement from these systems is rapid (Harris *et al.*, 1994) and the residence time in any buffer will therefore be very small. However, recent research (Harris *et al.*, 1996) has suggested that the combination of mole drainage and secondary drainage treatments in clay soils may, in effect, be overdraining the soil profile, and a lower drainage standard might be acceptable. Such systems, which restrict drainage until the soil profile is saturated, will increase the opportunities for adsorption of pesticides and the potential for buffers to be effective. Harris *et al.* (1996) found a reduction in loss of isoproturon of around 25% when drainage was restricted to periods of soil saturation.

Spray drift

Although primarily interested in the effects of buffers on bank vegetation, Marrs *et al.* (1993) and Marrs and Frost (1995) assessed spray drift travel and suggested that, on sites where seedling establishment was an important mechanism for community regeneration, buffer zones between farmland and the sites need to be 20 m. Seedlings of some species were affected at greater distances than established plants, indicating either greater capture of drift, or a greater sensitivity. This would clearly have implications where buffers were being established as part of an approach to create wetland flora, and could have a further consequence where floral survival might further influence contaminant removal.

Evidence of the effect of buffers on spray drift for features including hedges is provided by Davis *et al.* (1994) who demonstrated that hedges planted alongside water courses would affect the spray drift. They found that spray deposition immediately behind the hedge was substantially decreased but that this increased again up to 15 m away; in their studies this was nine times the height of the hedge. When tested against MCPA sprays, they found that the protection afforded by the hedge could be limited in strong winds. Similarly, Greig-Smith *et al.* (1992) suggested that hedges provided little resistance to drift when free of leaves at the end of winter.

IMPACT OF PESTICIDES ON THE BUFFER ZONE HABITAT

Damage to invertebrate populations following the spraying of insecticides is well documented and described in the United Kingdom by Muirhead-Thompson (1978), Crossland *et al.* (1982) and Pinder *et al.* (1993). The UK MAFF Pesticides Safety Directorate currently imposes 'no spray zone' label restrictions on more than 160 agrochemical products in order to prevent pesticides that represent a high risk to aquatic life from entering surface waters or ditches. These restrictions are designed to restrict spray drift and do not prevent the loss of pesticides to surface waters through drain flow or surface runoff which can occur at concentrations sufficiently high to pose a threat to aquatic life (Harris *et al.,* 1996; Williams *et al.,* 1996).

Quantification of the effects of pesticides in surface waters can be difficult, since natural population fluctuations may be extremely variable. Fleming *et al.* (1995) describe a freshwater mussel 'die-off' where some 1000 mussels were found to be dead or moribund in North Carolina in a stream whose catchment was dominated by forestry and agriculture. The event was suspected to be of acute origin since there was no evidence of disease and laboratory investigations detected cholinesterase inhibition. The authors suspect carbamate or organophosphate poisoning, but the cause could not be confirmed since there were no detectable levels of anticholinesterase pesticides in either mussel tissue or water samples.

Pollution events such as this are likely to be transient and thus difficult to detect, but are most likely to occur following heavy rainfall shortly after pesticide application. For example, Williams *et al.* (1996) demonstrated that rainfall events can generate transient pesticide concentrations in headwater streams that are fatal to *Gammarus pulex,* a freshwater shrimp. An *in-situ* bioassay (Matthiesson *et al.,* 1995) showed that following heavy rainfall peak concentrations of 27 μg/l carbofuran in a stream caused a cessation of feeding and 100% mortality of the test animals. Subsequent deployment of the *Gammarus pulex* bioassay following an application of chlorpyrifos led to mortality of 36% of test animals following a 12 mm rainfall event in 24 hours (47% of animals in the same test were assessed as moribund). The report concludes that transiently high pesticide concentrations are potentially significant to the ecology of streams and recommends that headwaters should be protected.

In addition to acute toxicity to stream invertebrates or fish, indirect effects may be caused by herbicide applications that deplete algal and macrophyte populations and deprive invertebrates of food (Hamala and Kollig, 1985). Forster and Botham (1996) measured the laboratory toxicity of trifluralin in three sediments (E_rC_{50} – 50% growth inhibition over 14 days) to *Lemna minor,* and compared it with measured concentrations of trifluralin in the stream. The maximum measured concentration was 50 μg/l and the lowest E_rC_{50} was found to be 0.4 mg/l. Although this represents a twentyfold difference and the minimum E_rC_{50} would be unlikely to cause damage to aquatic plants as a single dose, repeated inputs could reduce populations in the long term.

The relatively low toxicity of trifluralin to *Lemna minor* may be influenced by reduced bioavailability of the pesticide in the presence of sediment. Trifluralin binds strongly to organic matter and is not very soluble in water. Atrazine (a more mobile and soluble material) is an effective inhibitor of photosynthesis in algae and can affect algal communities in artificial ponds and streams at concentrations between 10 and 20 μg/l (Stratton, 1984). Despite this high inherent toxicity, there is evidence that algae can develop a tolerance to atrazine in agricultural streams (Hersh and Crumpton, 1988).

In recent years there has been increasing interest in the effects of pesticide pollution on benthic invertebrates in addition to the surface dwelling insects such as pond skaters and whirligig beetles that are most readily affected by overspray and spray drift. Particulate pesticide transport via drainflow or surface runoff is less likely than spray drift to affect surface dwelling and water column invertebrates, but benthic organisms may be at risk via ingestion of particulate material as well as exposure to interstitial water and the water column.

The increased use of no-spray zones and specially constructed buffer strips to reduce spray drift and leaching via drain flow and surface run-off will help to reduce pesticide contamination of both relatively impoverished headwater streams and, more importantly, biologically diverse surface waters. Some concern has been expressed at the potential build up of pesticides in such buffers and the consequent effect on the flora and fauna within the buffer area itself. This is of concern because data are difficult to obtain due to the natural variation in soil microbial populations (Edwards *et al.*, 1994). In a long-term study, Bromilow *et al.* (1996) found no deleterious effects on soil microbial populations after up to 20 years of applications of five pesticides to replicated plots on a silty clay loam soil, whereas Edwards *et al.* (1994), found reductions in microbial activity following the application of the fungicide captan in discrete microcosm studies.

DISCUSSION AND CONCLUSIONS

Although there are numerous reports citing the benefits that buffers can bring to reducing nutrient pollution, and suggesting design standards, papers citing procedures for the establishment of buffers to reduce pesticide contamination are relatively rare, indicating that this is an area where further research is needed. The lack of adequate guidance may be, in part, because the understanding of the mechanisms of pesticide transport is relatively new, compared to nitrate. In particular, the chemical properties of the pesticides themselves differ greatly and properties such as mobility and adsorption will greatly affect their travel paths and propensity to leach (Marshall *et al.*, 1996; Jones *et al.*, 1995).

The reported evidence suggests that relatively narrow buffers, between 2 and 10 m in width, are sufficient to trap both surface runoff particulates in many hillslope situations and aid the infiltration of surface water into the buffer (Baker *et al.*, 1995). However, equally important is that considerable pesticide movement is observed in sub-surface drainage in clay-based soils, entering the stream system effectively as point sources (Harris *et al.*, 1994). In such circumstances, surface buffers are unlikely to result in any noticeable reduction in pesticide movement to surface waters and buffers must be considered as three dimensional features.

For buffers to be effective in any situation it is evident therefore that the residence time is the most important variable for water quality improvement, since pesticide adsorption takes place only while the material is in the buffer. Where underdrainage is a problem, an approach showing some promise, and reported by Harris *et al.* (1996), has involved restricting drainage and placing absorbent materials within the drainage system to reduce pesticide loss. Depending on field slopes, there may be opportunities for extending this approach into three-dimensional streamside buffers, thus increasing wetness and residence time within the buffer, and thus providing opportunities to reduce pesticide contamination. Whereas a number of workers have assessed the hydrological and other limiting factors controlling nitrate removal in buffers (Haycock and Burt, 1993; Baker and Maltby, 1995) there is little published on the ideal hydrological conditions for pesticide removal. If pesticides have been successfully retained within a buffer, then degradation will be increased by increased soil moisture and temperature (Nicholls *et al.*, 1993) and may be further influenced by pH (Cox *et al.*, 1996). Also if increased pesticide adsorption in the buffer is a problem, then rapid degradation may be possible by the use of microbes with an enhanced degradation capability. For example, Cox *et al.* (1996) have identified and cultured a field soil which can degrade the herbicide isoproturon within a few days, compared to a normal half-life of around two months in the topsoil.

In addition to buffer width and the issue of sub-surface control of pesticide contamination, there are other features which need to be considered during buffer design. Buffers, as landscape features, can clearly influence spray drift essentially by removing the application of spray from this area. The wider the buffer, the greater the benefit, although widths up to 15 m might be necessary to eliminate most of the effects of spray drift (Davis *et al.*, 1994). Greig-Smith *et al.* (1992) suggested that open hedges had little effect on drift, indicating that further research is needed to maximise the design of the buffer to minimise drift, especially where economics dictate that only narrow buffers are feasible.

Buffers are generally considered as linear streamside features. In most situations the stream margin is the most appropriate location, although to control particulate movement, targeting the source may be preferable and this may be on the hillslope rather than stream margin area. The presence of considerable particulate loads in some drainage systems (Marshall *et al.*, 1996) provides evidence that the products of hillslope erosional activity can enter the drainage system, and thus bypass any streamside buffer. Equally, water and contaminants can enter streams as point sources, influenced by micro-topographical features. Linear buffers are therefore more suited to conservation than as a means of reducing contamination of watercourses. In a review of the role of buffers to control contamination of surface waters, Muscutt *et al.* (1993) concluded that targeted buffers are essential, and should be designed accordingly.

In conclusion, there is still much to learn about how buffers can be used to reduce pesticide contamination of surface waters. In particular, research is needed to determine a best practical approach, to include risk management considerations, and to cover the range of geographical situations and modes of transport for pesticides, so that those who advocate the use of buffers as landscape features, and those responsible for their implementation, can design effective systems.

REFERENCES

Asmussen, L.E., White, A.W., Hauser, E.W. and Sheridan, J.M. (1977) Reduction of 2,4-D load in surface runoff down a grassed waterway. *Journal of Environmental Quality,* **6** (2), 159-162.

Baker, C.J and Maltby, E. (1995) Nitrate removal by river marginal wetlands: Factors affecting the provision of a suitable denitrification environment. In: Hughes, J.M.R. and Heathwaite, A.L. (Eds). *Hydrology and Hydrochemistry of British Wetlands.* Published John Wiley & Sons Ltd, 291-313.

Baker, J.L., Mickleson, S.K., Hatfield, J.L., Fawcett, R.S., Hoffman, D.W., Franti, T.G., Peter, C.J. and Tierney, D.P. (1995) Reducing herbicide runoff: Role of best management practices. *BCPC Brighton Crop Protection Conference, Weeds,* **2**, 479-487.

Bengston, R.L., Southwick, L.M., Willis, G.H. and Carter, G.E. (1990) The influence of sub-surface drainage practices on herbicide losses. *Transactions American Society of Agricultural Engineering,* **32** (2), 415-418.

Beven, K.J. and Germann, P.F. (1982) Macropores and water flow in soils. *Water Resources Research,* **18**, 1311-1325.

Bromilow, R.H., Evans, A.A., Nicholls, P.H., Todd, A.D. and Briggs, G.G. (1996). The effects of soil fertility of repeated applications of pesticides over 20 years. *Proceeding COST 66 Workshop,* Stratford on Avon, 13-15 May 1996, 153-154.

Brown, C.D., Rose, D.A., Syers, J.K. and Hodgkinson, R.A. (1995) Effects of preferential flow upon the movement of pesticides and a conservative tracer from a heavy clay soil. Symposium at the University of Warwick, Coventry, 3-5 April. *BCPC Monograph No 62: Pesticide movement to water,* 93-98.

Buttle, J.M. (1990) Metachlor transport in surface runoff. *Journal of Environmental Quality,* **19**, 531-538.

Cox, L., Walker, A., Roberts, S.J. and Welch, S.J. (1996) Evidence for enhanced biodegradation of isoproturon in soils. *Proceeding COST 66 Workshop*, Stratford on Avon, 13-15 May 1996, 105-106.

Crossland, N.O., Shires, S.W. and Bennett, D. (1982) Aquatic toxicology of cypermethrin – III. Fate and biological effects of spray drift deposits in fresh water adjacent to agricultural land. *Aquatic Toxicology,* **2**, 253-270.

Davis, B.N.K., Frost, M.J., Yates, A.J. and Plant, R.A. (1994) The effects of hedges on spray deposition and on the biological impact of pesticide spray drift. *Ecotoxicology and Environmental Safety,* **27** (3), 281-293.

Dillaha, T.A. (1989) *Water quality impacts of vegetative filter strips.* ASAE Paper No 89-2043. American Society Agricultural Engineering, St Joseph, MI.
Dillaha, T.A., Reneau, R.B., Mostaghimi, S., Shanholtz, V.O. and Magette, W.L. (1987) *Evaluating nutrient and sediment losses from agricultural lands: vegetated filter strips.* US Environmental Protection Agency Report No. CBP/TRS 2/87, Washington DC.
Edwards, C.A., Subler, S. and Chen, S-K. (1994) A microcosm system for evaluating the effects of pesticides on dynamic soil processes and soil organisms. *BCPC Brighton Crop Protection Conference, Pests and Diseases,* **3**, 1301-1306.
Fennesey, S. (1993) *Riparian buffer strips – their effectiveness for the control of agricultural pollution.* World Wide Fund for Nature Project 210/89. Wetlands and Nutrient Stripping, 79pp.
Fleming, W.J., Augspurger, T.P. and Alderman, J.A. (1995) Freshwater mussel die-off attributed to anticholinesterase poisoning. *Environmental Toxicology and Chemistry,* **14**, 877-879.
Forster, A. and Botham, K. (1996). Relationships between pesticide movement to surface water and toxic effects in mesocosms and microcosms. *Proceedings of COST 66 Workshop,* Stratford on Avon, 13-15 May 1996, 181-183.
Ganzelmeier, H. (1993) *Criteria for assessment of plant protection products in the registration procedure* (1993). BBA Braunschweig, Germany.
Gillet, H. (1991) *Contamination des eaux par les pesticides en Bretagne. Etude d'e(E) valuation par bassins versants.* SPRV – DRAF. 7pp
Greig-Smith, P., Eagle, D., Williams, C.T. and Brown, P.M. (1992) Distribution and persistence of sprayed pesticides and residues in water. In: Greig-Smith, P., Frampton, G. and Hardy, T. (Eds.) *Pesticides, Cereal Farming and the Environment, The Boxworth Project.* HMSO, 60-67.
Hamala, J.A. and Kollig, H.P. (1985) The effects of atrazine on periphyton communities in controlled laboratory ecosystems. *Chemosphere,* **14**, 1391-1408.
Harris, G.L., Turnbull, A.B., Gilbert, A.J., Christian, D.G. and Mason, D.J. (1992) Pesticide application and deposition – their importance for leaching to surface waters. *BCPC Brighton Crop Protection Conference, Pests and Diseases,* **3**, 477-486.
Harris, G.L., Bailey, S.W., Rose, S.C., Mason, D.J. and Llewellyn, N. (1993) The transport of pesticide residues to surface waters in small clay-based catchments. *BCPC Brighton Crop Protection Conference, Weeds,* **2**, 815-820.
Harris, G.L., Nicholls, P.H., Bailey, S.W., Howse, K.R. and Mason, D.J. (1994) Factors influencing the loss of pesticides in drainage from a cracking clay soil. *Journal of Hydrology,* **159**, 235-253.
Harris, G.L. (1995) Pesticide loss to water – a review of possible agricultural management opportunities to minimise pesticide movement. Symposium at the University of Warwick, Coventry, 3-5 April. *BCPC Monograph No 62: Pesticide movement to water,* 3711-380.
Harris, G.L., Jones, R.L., Catt, J.A., Mason, D.J. and Arnold, D.J. (1996) Behaviour of pesticides in cracking clays. *Proceeding COST 66 Workshop,* Stratford on Avon, 13-15 May 1996, 193-194.
Haycock, N.E. and Burt, T.P. (1993) Role of floodplain sediments in reducing the nitrate concentrations of subsurface runoff: a case study in the Cotswolds. *Hydrological Processes,* **7**, 287-295.
Hersh, C.M. and Crumpton, W.G. (1988) Atrazine tolerance of algae isolated from two agricultural streams. *Environmental Toxicology and Chemistry,* **8**, 333-338.
House, W.A., Rae, J.E. and Kimblin, R.T. (1992) Source-sediment controls on riverine transport of pesticides. *BCPC Brighton Crop Protection Conference, Pests and Diseases,* **2**, 865-870.
Jones, R.L. (1993) Role of field studies in assessing environmental behaviour of herbicides. *BCPC Brighton Crop Protection Conference, Weeds,* **3**, 1275-1282.
Jones, R.L., Harris, G.L., Catt, J.A., Bromilow, R.H., Mason, D.J. and Arnold, D.J. (1995) Management practices for reducing movement of pesticides to surface water in cracking clay soils. *BCPC Brighton Crop Protection Conference, Weeds,* **2**, 489-498.
Keskitalo, J. (1990) Occurrence of vegetated buffer zones along brooks in the catchment area of lake Tuusulanjaπrvi, south Finland. *Aqua Fennica,* **20** (1), 55-64.
Leonard, R.A. (1988) Herbicides in surface waters. In: Grover, R. (Ed.) *Environmental Chemistry of Herbicides,* CRC Press, pp 45-87.
Leonard, R.A. (1990) Movement of pesticides into surface water. In: Cheng, H.H. (Ed.) *Pesticides in the Soil Environment: Processes, impacts and modelling.* SSA Book Series 2. Madison, WI. Soil Science Society of America, 303-349.
Lloyd, G.A. and Bell, G.J. (1993) *Hydraulic nozzles: Comparative spray drift study.* Report of a study carried out in 1983 in association with the BAA.
Marrs, R.H., Frost, A.J., Plant, R.A. and Lunnis, P. (1993) Determination of buffer zones to protect seedlings of non-target plants from the effects of glyphosate spray drift. *Agriculture, Ecosystems and Environment,* **45**, 283-293.
Marrs, R.H. and Frost, A.J. (1995) Minimising the effects of herbicide spray drift. *Pesticide Outlook,* 28-31.
Marshall, V.L., Hollis, J., Brown, C.D., Forster, A., Harris, G.L., Mason, D.J. and Farr, I. (1996) Quantifying the potential for particulate transport of pesticides from eight representative UK soil types. *X Symposium Pesticide Chemistry,* Piazence, Italy, Sep-Oct 1996.

Matthiessen, P., Sheahan, D., Harrison, R., Kirby, M., Rycroft, R., Turnbull, A., Volkner, C. and Williams, R. (1995) Use of a *Gammarus pulex* bioassay to measure the effects of transient carbofuran runoff from farmland. *Ecotoxicology and Environmental Safety,* **30**, 111-119.

Mickleson, S.K. and Baker, J.L. (1993) Buffer strips for controlling herbicide runoff losses. *ASAE/CSAE Meeting Presentation Paper* No 93-2084. Spokane Centre, Spokane, Washington, 20-23 June, 1993.

Monke, E.J., Kladivko, E.J., van Scoyoc, G.E., Martinez, M. and Huffman, R.L. (1989) Movement of pesticides and nutrients in drainage water. In: Dodd, V.A. and Grace, P.M. (Eds.) *Land and Water Use,* Balkema, Rotterdam, 297-304.

Muirhead-Thomson, R.C. (1978) Lethal and behavioural impact of chlorpyrifos methyl and temephos on select stream macroinvertebrates: experimental studies on downstream drift. *Archives of Environmental Contamination and Toxicology,* **7**, 139-147.

Muscutt, A.D., Harris, G.L., Bailey, S.W. and Davies, D.B. (1993) Buffer zones to improve water quality: a review of their potential use in UK agriculture. *Agriculture, Ecosystems and Environment*, 59-77.

Neely, R.K. and Baker, J.L. (1989) Nitrogen and phosphorus dynamics and the fate of agricultural runoff. In: Van der Walk (Ed.) *Northern Prairie Wetlands, I*owa State University Press, 92-131.

Nicholls, P.H., Evans, A.A., Bromilow, R.H., Howse, K.R., Harris, G.L., Rose, S.C., Pepper, T.J. and Mason, D.J. (1993) Persistence and leaching of isoproturon and mecoprop in the Brimstone Farm plots. *BCPC Brighton Crop Protection Conference, Weeds,* **2**, 849-854.

Niebling, W.H. and Alberts, E.E. (1979) *Composition and yield of soil particles transported through sod strips.* ASAE paper No 79-2065. American Society of Agricultural Engineering, St Joseph, MI.

Norris, V. (1993) The use of buffer zones to protect water quality. *Water Resources Management,* **7**, 257-272.

Parsons, J.E., Daniels, R.D., Gilliam, J.W. and Dillaha, T.A. (1990) *Water quality impacts of vegetative filter strips.* ASAE Paper No. 90-2501. American Society of Agricultural Engineering, St Joseph, MI.

Patty, L., Gril, J.J., Real, B. and Guyot, C. (1995) Grassed buffer strips to reduce herbicide concentration in runoff. Preliminary study in western France. Symposium at the University of Warwick, Coventry, 3-5 April. *BCPC Monograph No 62: Pesticide movement to water,* 397-406.

Pinder, L.C.V., House, W.A. and Farr, I.S. (1993) Effects of insecticides on freshwater invertebrates. In: Cooke, A.S. (Ed.) *Environmental effects of pesticide drift.* English Nature.

Smith, C.M. (1989) Riparian pasture retirement effects on sediment, phosphorus and nitrogen in channelised surface run-off from pastures. *New Zealand Journal of Freshwater Research,* **23**, 139-146.

Speller, C.S., Cleal, R.A.E. and Runham, S.R. (1992) A comparison of winter wheat yields from headlands with other positions in five fen peat fields. *BCPC Monograph No 50. Set-aside,* 47-50.

Stratton, G.W. (1984) Effects of the herbicide atrazine and its degradation products, alone and in combination, on phototrophic microorganisms. *Archives of Environmental Contamination and Toxicology,* **13**, 35-42.

Traub-Eberhard, U., Henchel, K., Koördel, W., Klein, W. (1995). Influence of different field sites on pesticide movement into subsurface drains. *Pesticide Science,* **43**, 121-129.

Wauchope, R.D. (1978) The pesticide content of surface water draining from agricultural fields. *Journal of Environmental Quality,* **7**, 459-472.

Wauchope, R.D. and Decoursey. (1986) Measuring and predicting losses of herbicides in runoff water from agricultural fields. In: Camper, N.D. (Ed.) *Research Methods in Weed Science, 3rd Edition.* Southern Weed Society, 135-254.

White, S.L. and Pinkstone, D.C. (1995) The occurrence of pesticides in drinking water. *British Crop Protection Council Monograph No 62: Pesticide movement to water,* 263-268.

Williams, R.J., Brook, D.N., Glendinning, P.J., Matthiessen, P., Mills, M.J. and Turnbull, A. (1991). Movement and modelling of pesticide residues at Rosemaund farm. *BCPC Brighton Crop Protection Conference, Weeds,* **2**, 507-514.

Williams, R.J., Brooke, D.N., Clare, R.W., Matthiessen, P. and Mitchell R.D.J. (1996) *Report No. 129 – Rosemaund Pesticide Transport Study 1987-1993.* Institute of Hydrology, Wallingford, Oxon, UK.

Worral, F., Parker, A. and Rae, J.E. (1993) Suspended and colloidal matter in leachate from lysimeters. Implications for pesticide transport and lysimeter studies. *BCPC Brighton Crop Protection Conference, Weeds*, **3**, 899-903.

8 Grassed buffer zones to limit contamination of surface waters by pesticides: research and action in France

J.J. Gril[1], B. Real[2], L. Patty[3], M. Fagot[4] and I. Perret[5]

[1]*Institut de recherche pour l'ingéniérie de l'agriculture et de l'environnement (Cemagref), Water Quality Division, Lyon, France*
[2]*Institut Technique des Céréales et des Fourrages (ITCF), Estrées-Mons, France*
[3]*Institut Technique des Céréales et des Fourrages, Lyon, France*
[4]*Ministry of Environment, Water Directorate, Paris, France*
[5]*Ministry of Agriculture, Rural Areas and Forestry Directorate, Paris, France*

Abstract

Over the past few years, monitoring of water tables and surface freshwater in France has shown significant cases of pesticide contamination and many surface water samples have exceeded the set limits for drinking water (0.1 μg l^{-1} European Directive 80/778/CEE). As a consequence public authorities, in collaboration with agricultural and rural development partners, have decided to initiate activities to limit this form of contamination.

Contamination is limited firstly by the improvement of cultivation methods: control of runoff and erosion, choice of pesticides, reduction of application rates and amelioration of spraying. The scientific bases of these methods are understood quite well and farmers simply require information and training to implement them. The CORPEN (see note) working group has specific responsibilities to circulate technical data amongst local organisations. However, the implementation of these approaches are difficult to control and they are therefore insufficient on their own.

The second way of limiting contamination is through the filtration of runoff before it joins the water networks. There appears to be widespread demand for an increased use of buffer zones and in particular grass buffer strips. Therefore, there is an immediate need to establish design criteria and identify appropriate locations for these buffers within the catchment. It is also important to avoid inappropriate applications of the buffer zone approach which, over the long term, would bring adverse publicity to this promising method. However, studies are quite rare and it is difficult at present to establish design criteria on the basis of current scientific knowledge. For this reason, a two-tier approach has been set up in France. Firstly, research projects look at the effectiveness of grass buffer designs and also at the direction of water circulation and location of buffers. Secondly, the CORPEN working group has been set up to collect and synthesise present knowledge (concerning hydraulics, effectiveness and choice of plant species) in order to avoid basic errors being repeated.

INTRODUCTION

The use of grassed buffer zones (GBZs) to reduce pesticide transfer to surface waters has aroused a great interest in France recently, triggered by drinking-water regulations and sustainable agriculture programs. This particular best management practice often causes reactions which are more emotional than scientific: many are in favour of GBZs, others not. In fact, GBZs are neither a universal panacea, nor totally inappropriate – just as it could be said for all other crop protection practices (choice of active ingredients, application rates, mechanical weed control, etc.). On the one hand a widespread use of GBZs without effective results could lead to an abandonment of this practice in the long term whilst on the other hand, a simple rejection of the technique's potential would also put an end to a promising way to improve surface water quality.

The "CORPEN" is currently assessing what information is available on the principles and practice of

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0

using GBZs and will soon publish a practical guide. Other elements are subjects for research programs (ongoing or planned).

We have identified three series of questions which require answers in relation to the appropriate design and implementation of GBZs:

- To what extent are GBZs effective in restricting pesticide transfer in runoff? Are we able to model both the GBZ and the characteristics of runoff?
- Where should GBZs be located in the watershed: along rivers, as a grass strip downslope of the fields or in hill-slope hollows where concentrated runoff collects?
- Are we able to give practical recommendations which are both technically and socioeconomically sound for GBZ implementation and maintenance?

Our paper reports results relating to the first questions.

EFFECTIVENESS OF GBZs

GBZs have been studied and used since 1965, mostly in the U.S., to reduce erosion and nutrient transfer in runoff. Historical data show that buffer strips increase water infiltration, reduce nutrient transport from feedlots and trap suspended solids via filtration and sedimentation in the strips. As pesticides in runoff are both in a soluble form and a particulate form (adsorbed to fine clay particles and organic matter), it was not obvious that GBZs would reduce pesticide transfer in runoff in addition to limiting erosion. Asmussen *et al.* (1977) and Rhode *et al.* (1980) reported that a 24.4 m grassed waterway reduced 2,4 D (soluble) and trifluralin (strongly adsorbed particulate) losses in runoff by an average 70 and 94%, respectively.

In France, studies have been conducted since 1992 by ITCF and Cemagref (in collaboration with Rhône-Poulenc Agro and the Ministry of Agriculture) to evaluate the effectiveness of GBZs in restricting pesticide transfer in runoff. Pesticides with different environmental behaviours were selected (Table 1).

Table 1. Physico-chemical properties of pesticides applied

	IPU	*DFF*	*lindane*	*atrazine*
Water solubility (mg $litre^{-1}$)[a]	65	0.05	7	33
K_{oc} ($cm^3\,g^{-1}$)[a]	120	1990	1100	100
half-life (days)[a]	12-32	175-294	100-120	60-70

[a] : from Rhône-Poulenc Agro
K_{oc} : Organic carbon partition coefficient

The first study, started in 1992 at ITCF's La Jaillière experimental farm in Brittany, aimed to determine the effectiveness of a 5.7 m and a 11.1 m GBZ in reducing isoproturon (IPU) and diflufenican (DFF) transfer in runoff generated on small plots (125 m^2). Since 1993, additional experimental sites were implemented at ITCF's La Jaillière, Bignan and Plélo research farms in Brittany for two purposes: to assess the effectiveness of 6, 12 and 18 m grassed strips at reducing lindane, atrazine and its metabolites and to confirm preliminary results obtained with IPU and DFF in a range of soil and cropping conditions. Runoff in these experiments was generated from larger cultivated plots (250 m^2).

Variations in runoff volume and residue concentration observed in the different experimental conditions provide an overview of GBZ effectiveness (Patty *et al.*, 1995a). Runoff volume was reduced by 8 to 99.9% within the GBZs and 69 to 100% of suspended solids were retained. Lindane and atrazine were reduced by 72 to 100% and by 44 to 100% respectively. The atrazine metabolites were reduced by 45 to 100% within the GBZs. Finally, IPU and DFF were reduced by 75 to 99% and by 68 to 97% respectively. GBZs were shown to be effective in reducing pesticide transfer in runoff under various experimental conditions, during the whole cropping period, including the first runoff events following pesticide application (Figure 1). GBZs effectiveness seems to be independent of rainfall intensity – at least, in these particular experimental conditions. Moreover, in spite of experimental limitations, rainfall simulation results obtained in 1995 at La Jaillière showed that the strips were still effective in conditions of intense runoff (Figure 2).

Figure 1. IPU (a) and DFF (b) losses in runoff at Plélo (1994-95 cropping period)
[B6: Strip width = 6 m; B12: Strip width = 12 m, etc.]

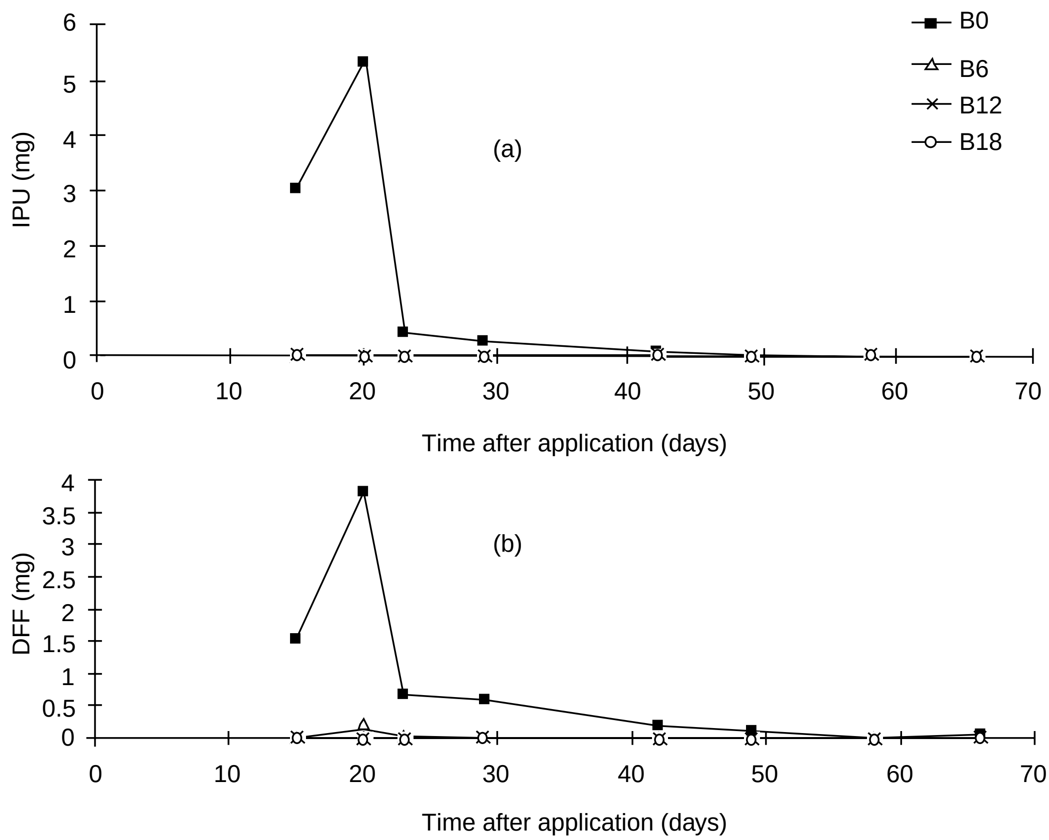
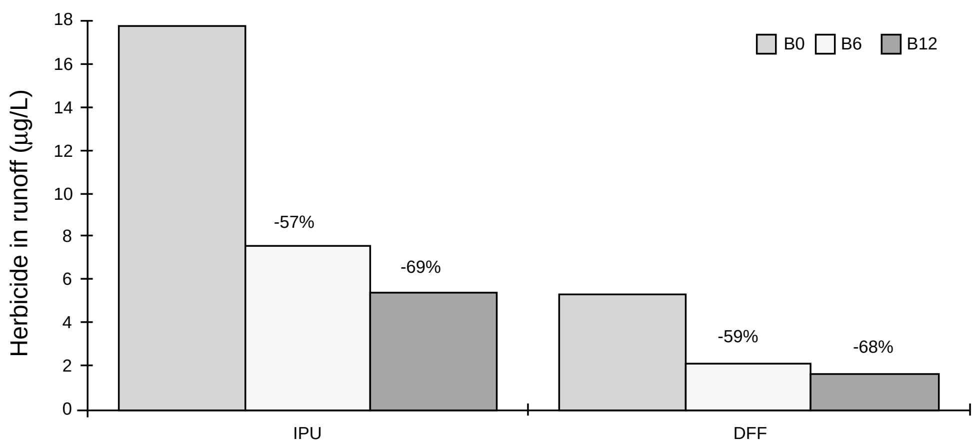


Figure 2. IPU and DFF concentrations in runoff at La Jaillière (1995 rainfall simulation)



It is clear that a number of hydrological, physical and chemical processes are involved in the functioning of GBZs including infiltration of water and soluble pollutants within the strips and the retention of sediment-bound pollutants due to filtration and sedimentation. Preliminary results obtained at La Jaillière showed that IPU and DFF concentrations in runoff were depleted within the strips. As infiltration and sedimentation are not likely to induce the large reductions observed in herbicide concentrations, we assume that sorption of pesticides onto organic matter and vegetation in the GBZ is a significant factor in their effectiveness. But this assumption has to be verified with further experiments.

Our results are in agreement with recent literature (Baker *et al.*, 1995) and show that GBZs provide a way to improve surface water quality in agricultural areas. But there are still many questions to be answered. For example, do residues accumulate in the strip or leach through the soil and reach groundwater? Baker *et al.* (1995) reported that atrazine and cyanazine concentrations declined in a buffer strip as the season progressed, presumably due to degradation.

These experiments demonstrate the effectiveness of GBZs. However, work should continue to evaluate the long term effects under various conditions. Current results are not yet sufficient to make conclusions about adequate GBZ widths. In particular, there is a need for more data on the interception of concentrated runoff within GBZs.

Runoff simulation experiments have also been undertaken by Cemagref and ITCF as a tool to test a variety of situations (inlet flow rates, density of grass coverage, GBZ slope, etc.). The influence of the amount of grass residue on the GBZ is also being observed. These experiments are still in progress.

Note
CORPEN – "Comité d'orientation pour la réduction de la pollution des eaux par les nitrates, les phosphates et les produits phytosanitaires provenant de l'activité agricole" : committee working on the reduction of water pollution caused by nitrates, phosphates and pesticides: this collaborative organisation gathers specialists from public authorities and private organisations involved in agriculture and water protection. The CORPEN has already published general guidelines to optimise chemical treatments for crop protection and methodological guidelines to diagnose causes of contamination in a watershed. Documents concerning GBZs and good spraying methods will be published in the near future.

REFERENCES

Asmussen, L.E., White, A.W., Hauser, E.W. and Sheridan, J.M. (1977) Reduction of 2,4 D load in surface runoff down a grassed waterway. *J. Environ. Qual.*, **6**, 159-162.

Baker, J.L., Mickelson, S.K., Hatfield, J.L., Fawcett, R.S., Hoffman, D.W., Franti, T.G., Peter, C.J. and Tierney, D.P. (1995) *Reducing herbicide runoff: role of best management practices.* Brighton Crop Protection Conference – Weeds, 479-487.

Patty, L., Gril, J.J., Réal, B. and Guyot, C., (1995a) *Grassed buffer strips to reduce herbicide concentration in runoff – Preliminary study in western France.* Symposium at the University of Warwick (U.K), Coventry 3-5 April 1995, BCPC Monograph 62: Pesticide movement to water, 397-406.

Patty, L., Réal, B. and Masson, E. (1995b) *Limitation du transfert de l'atrazine et du lindane par ruissellement – Efficacité des bandes enherbées.* ANPP – Sixteenth COLUMA conference – International meeting on weed control, Reims, 6-7-8 December, I/III , 75-82.

Rhode, W.A., Asmussen, L.E., Hauser, E.W., Wauchope, R.D. and Allison, H.D. (1980) Trifluralin movement in runoff from a small agricultural watershed. *J. Environ. Qual.*, **9**, 37-42.

9 The relationship between land use, river bank quality and lotic macroinvertebrate community

B. Gumiero and G. Salmoiraghi

Department of Biology, University of Bologna, Via Selmi. 3, Bologna, 40126, Italy.

Abstract

In the last few years a lot of concern has been given to riparian zones as terrestrial-wetland ecotones, able to control the nutrients moving from the drainage basin into the river. But riparian zones also constitute an important water-wetland ecotone which influences many environmental parameters in the river such as temperature, radiation etc. and represents a vital trophic resource for the benthic communities. An important but still unanswered question is whether these riparian zones, strongly modified at their terrestrial-wetland ecotone through an increase of nutrient loads from the drainage basin, can affect their water-wetland ecotone and, in turn, the adjacent benthic communities.

In this study we evaluated in 64 different riparian zones how nearby land use could affect the riparian structure and, in turn, the river benthic biodiversity. We used two new environmental indices developed in the last few years by a multidisciplinary research group (Braioni *et al.*, 1995) to assess river bank quality: 1) The natural state index which reflects the state of nature conservation on the bank and the potentiality of the area to support a high level of biodiversity, 2) The buffer strip index which gives an evaluation of the capacity of the riparian zone to filter, metabolise and bioaccumulate nutrients and pollutants.

INTRODUCTION

A large number of studies have shown how a river responds to changes in water and sediment discharges which result from alterations in land use, such as the conversion of forest to agriculture and urbanisation (Brookes, 1988). In Italy there has been a massive transformation of the fluvial environment, particularly with regard to riparian vegetation due to intensive land use.

In the last few years, a lot of attention has been given to the ability of riparian zones, as terrestrial-wetland ecotones, to retain the nutrients moving from the drainage basin into the river (Pinay *et al.*, 1990). However, riparian zones also constitute important water-wetland ecotones which influence many environmental parameters in the river such as temperature, radiation etc. (Fisher, 1977) and represent a vital trophic resource for benthic communities (Anderson and Cummins, 1979; Cummins and Klung, 1979; Webster and Benfield, 1986). A co-evolutionary process, as described by Cummins *et al.* (1989), points out the link between riparian vegetation and macroinvertebrate communities.

An important, but still unanswered question, is whether these riparian zones, strongly modified at their terrestrial-wetland ecotones through an increase of nutrient loads from the drainage basin can affect their water-wetland ecotones and, in turn, the adjacent benthic communities.

The structure and function of biotic communities have been considered as an interrelated system to monitor changes which occur in the environment (Naiman and Décamps, 1990). Macroinvertebrates have been the most studied communities in flowing water environments (Hellawell, 1986; Rosenber and Resh, 1992).

The aim of our research is to study the relationship between the different uses of buffer zones and the neighbouring aquatic community, which requires the evaluation of the aquatic biodiversity while looking at the quality of the riparian zone. This tests the hypothesis that land use near the riparian zone affects the structure of the zone which, in turn, modifies the structure of the aquatic community.

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0

With this in mind we set up a preliminary investigation, on a basin scale, over a vast area (Reno River Basin) in the Bologna region between the Apennines and the Po River. A survey on land use and riparian zones, and evaluation of lotic macroinvertebrate biodiversity was carried out in 51 sites throughout the upper part of the Reno River Basin.

STUDY AREA

The Reno River is the ninth longest river in Italy (220 km) crossing two Regions: Tuscany and Emilia-Romagna. It ends in a catchment basin of 4953 km^2 with 1 million inhabitants (Fig. 1).

Figure 1. Map of the Reno River Basin. The circles show the sampling sites, the arrows and numbers show the discharge sections.

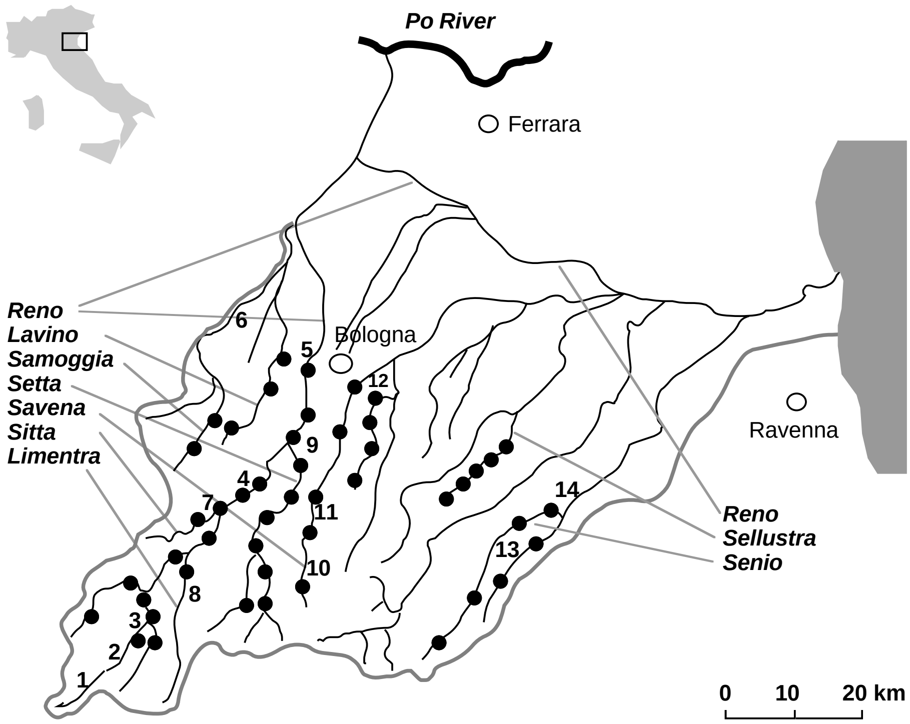


Figure 2. Seasonal patterns in rainfall and discharge at two sites on the Reno River. The site A is located in the upper part of the basin, section n°2 and the site B is located in the lower part of the area under study, section n° 5 (see Fig. 1). Data give the historical average over 50 years.

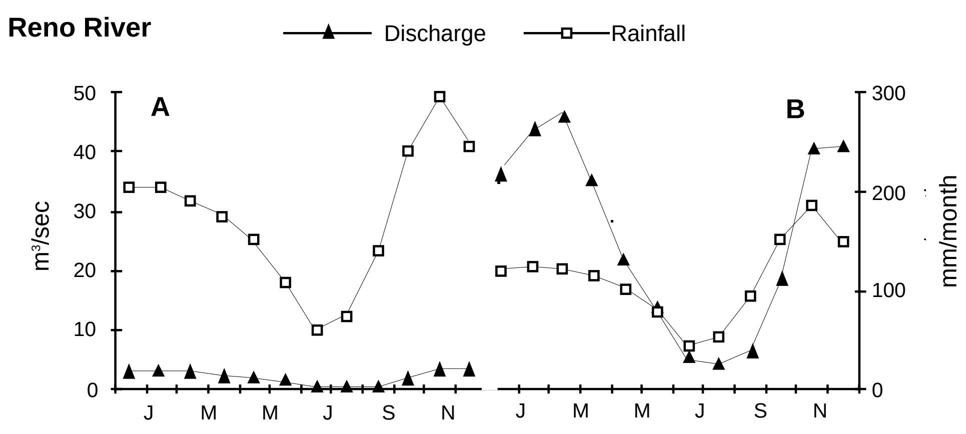


Table 1. Characteristics of the main streams in the study at different discharge sections. The section numbers are shown on the map of the study area.

Water courses	*Section number*	*Basin area (km^2)*	*Site altitude (m)*	*Average altitiude (m)*	*Mean discharge*
Reno	1	12.8	665	870	0.66
Reno	2	39.1	595	893	2.00
Reno	3	89.2	530	951	3.99
Reno	4	581	160	735	18.11
Reno	5	1051	60	639	25.5
Samoggia	6	170	44	375	2.1
Silla	7	81.4	330	873	3.05
Limentra	8	106.7	345	784	4.01
Setta	9	315.8	130	623	7.5
Savena	10	11.5	720	1005	0.38
Savena	11	77	325	752	1.67
Savena	12	157	72.9	530	2.36
Sintria	13	21	325	605	0.38
Senio	14	269	33	432	3.09

The Reno river and its tributaries (Table 1) are characterised by a medium rainfall of 700/800 mm per year and, as are all Apennine streams, with wide discharge fluctuations throughout the year. The typical hydrograph has peak discharge in spring and fall, and low flow in summer. Moreover, the discharge is often strongly affected by river regulation. The Northern Apennines river basin geology is characterised by two layers of easily degradable clastic sedimentary rocks (clay, sandstone, sand) (Cattaneo *et al.,* 1995). In the streams under study, impermeable basins are usually found in which temporal variations in runoff are less pronounced than rainfall variations (Fig. 2).

As is often the case elsewhere, the boundary between the hill region and the plain also divides the Reno River basin into two parts. The upper part is partially natural and partially modified while the lower part of the basin has been substantially affected by both land use and extensive modifications of the river course (channelisation). For these reasons our study focused only on the upper part of the basin.

METHODS

Qualitative kick samples of the macroinvertebrate community were collected in 51 sites, from 1993 to 1995 throughout the area under study in the upper part of Reno River Basin. The amount of time spent sampling each habitat type was proportional to the area it occupied. The samplings were carried out at least twice, once with low discharge (July) and again with medium discharge (March). At the same time inventory sheets of the area (Braioni *et al.,* 1994) were completed giving, in particular, details about the riparian zone and surrounding countryside. Furthermore, the features of the buffer strip and land use were integrated with careful observations of suitable maps: scale 1:25000, for riparian vegetation and land use.

The lotic macroinvertebrate biodiversity was assessed by the number of taxa and the Extended Biotic Index, originally proposed by Woodiwiss (1978), which has become widely used in Italy. In this investigation the modified Extended Biotic Index was used, in accordance with the application method suggested by Ghetti and Bonazzi (1981) and Ghetti (1986). This index is based on the presence/absence of key taxa whose tolerance to pollution is known, and on taxon richness. All taxa present are taken into account and different scores are given, depending on the total number of taxa and the presence of taxa sensitive to pollution. For this index, the macroinvertebrates of a stream reach are collected and classified to a standard level, genus or family, depending on the group. The range of E.B.I. values is from 13 to 1 and these values are divided into five classes of environmental quality from I (the best) to V (the worst).

The river bank quality was evaluated using two new environmental indices (Braioni *et al.*, 1994; Braioni, 1996): 1) The "wild state index" (W.S.I.) which reflects the state of nature conservation on the bank and the potential of the area to support a high level of biodiversity; and 2) the "buffer strip index" (B.S.I.) which gives an evaluation of the capacity of the riparian zone to filter, metabolise and bioaccumulate nutrients and pollutants.

The inventory sheet is the same for both indices. It is organised into 31 variables. 25 variables are used in the normal application of the indices. These variables relate to surrounding landscape, river bed, banks (e.g. angle, height, surface), floodplain, trees, shrubs, herbaceous vegetation, crops, buildings, railway tracks, embankments, diversion of flow, excavation, inflows, irrigation and other human impact. Each index is evaluated by a separate numerical weighting given to each of the variables on the inventory sheet. The range of values is from +8 to -10 for the "wild state index" and from +8 to -8 for the "buffer strip index". The final scores were divided into five classes of environmental quality from I (the best) to V (the worst). Both indices are based almost on the same parameters but the weight of each parameter differs in accordance to the aim of each index.

To classify the use of the land surrounding the site, an area of 4 km^2 around each station was surveyed and six different typologies were obtained: 1) Urban-Cultivated (UC); 2) Cultivated (C); 3) Cultivated-Urban-Natural (CUN); 4) Cultivated-Natural (CN); 5) Urban-Natural (UN) and 6) Natural (N).

An analysis of variance (ANOVA) was used to test the differences between land use, the two river bank indices and aquatic invertebrate biodiversity. Finally, to compare the total number of taxa and E.B.I. values with the scores of the two river bank quality indices, the Spearman Rank Correlation was used.

RESULTS AND DISCUSSION

The buffer zone in the upper and medium part of the basin is characterised by gravel soil and by a wide range of water levels.

The riparian vegetation in this area is, on the whole, rather homogeneous; poplars and willows being the most common. The intermediate stage between *Populus* and *Salix* is the most common type of vegetation. The most common species of tree is *Populus nigra* with a smaller number of *Populus canescen,* while *Salix purpurea, Salix eleagnos, Salix fragilis* and *Salix alba* are the most common shrubs. Some allochtonus and opportunistic species such as *Robinia pseudoacacia* are also abundant, especially in the more disturbed reaches. On the other hand, in the most pristine areas only a few examples of *Alnus glutinosa* are found and rarely in abundance (Table 2).

Away from the river this riparian vegetation is more usually replaced by a cultivated and urbanised area or road rather than woodland. In the agricultural areas permanent crops, arable crops, grassland and market gardens are cultivated.

Considering the "wild state index" values, the sites were classified into the five quality classes with most of them belonging in equal numbers to the second and third classes of "wildness" and only a few of them to the fourth. Meanwhile, the buffer capacity of these sites was evaluated using "buffer strip index". The distribution among the five classes of this index clearly indicates lower values compared to W.S.I.. Only 20% of the sites were classified into the second class of B.S.I., 60% into the third and 20% into the fourth.

The invertebrate communities are made up mainly of five insect groups: Ephemeroptera, Thricoptera, Plecoptera, Coleoptera and Diptera. The Plecoptera, the most sensitive to the environmental changes, was represented almost completely by the genera Leuctra, one of the most tolerant of this group. The most diversified group was the Ephemeroptera with 8 genera (Table 2).

In the basin under study the term natural was used to refer to those areas in the most untouched state, as there were no true pristine ones. As a result, no examples of the first quality class for each index were considered and few of the most sensitive macrozoobenthic were found.

Table 2. Main vegetation species and main macroinvertebrate taxa found in the streams under study.

Vegetation Species	*Reno*	*Sam.*	*Lav.*	*Silla*	*Ver.*	*Ven.*	*Sam.*	*Lim.*	*Setta*	*Sav.*	*Sell.*	*Sen.*	*Sint.*
Alnus glutinosa	*		*	*			*		*	*		*	*
Populus alba	*	*							*			*	*
Populus nigra	*	*	*	*			*	*		*	*	*	
Populus canescens	*	*	*		*	*					*	*	*
Salix alba	*	*	*			*			*		*	*	*
Ulmus campestris	*	*	*								*		*
Robinia pseudoacacia	*	*	*	*	*	*		*	*	*	*	*	*
Rubus ulmifolius	*	*	*	*				*	*			*	*
Sambucus nigra	*		*								*	*	*
Salix eleagnos	*			*		*	*	*	*	*	*	*	
Salix purpurea	*						*		*	*	*		
Salix viminalis	*	*								*		*	
MACROINV. TAXA													
Leuctra	*	*		*			*	*	*	*		*	*
Baetis	*	*	*	*	*	*	*	*	*	*	*	*	*
Caenis	*	*	*						*	*	*	*	*
Ecdyonurus	*			*		*	*	*	*	*		*	*
Habrophlebia	*	*		*		*	*	*	*	*		*	*
Hydropsychidae	*	*	*	*	*		*	*	*	*	*	*	*
Limnephilidae	*		*				*	*	*	*			
Rhyacophlidae	*			*			*	*	*	*		*	*
Elminthidae	*	*		*			*	*	*	*		*	*
Hydraenidae	*			*			*	*	*	*			*
Chironomidae	*	*	*	*	*	*	*	*	*	*	*	*	*
Simulidae	*	*	*	*	*			*	*	*	*		*

Considering that the riparian zone is both a terrestrial-wetland ecotone and also an important water-wetland ecotone, the ANOVA and the Spearman Rank Correlation were used to find the relationship between the three environments: river, riparian and surrounding area.

The stream riparian areas are not disturbed unless a large proportion of the catchment is devoted to agriculture. As the land devoted to agriculture increases, there is a much higher rate of loss of riparian stream-side vegetation. The ANOVA test shows highly significant ($p \leq 0.01$) differences between the different land uses under consideration with respect to the "wild state index" and the "buffer strip index" scores (Fig. 3). This points out the strong relationship between land use and buffer zone structure in the area under study.

When the quality classes of "wild state index" and "buffer strip index" were compared with respect to macroinvertebrate biodiversity, only the "wild state index" classes showed significant differences ($p \leq 0.05$) on both taxa and E.B.I. values (fig. 4). Moreover, the correlations between the two river bank index values and the macroinvertebrate biodiversity (EBI values and n° of taxa units) were tested by Spearman Rank Correlation. The values of "buffer strip index" and macroinvertebrate were not significantly correlated but the correlation was highly significant ($p \leq 0.01$) between "wild state index" and benthic organisms. This may indicate that the biodiversity of the invertebrate communities was more affected by the natural state of the riparian zone than its buffer functions.

Although this aspect needs to be verified through a specific investigation, the clear relationship between the wildness of riparian zones and the structure and function of the macroinvertebrate community was confirmed (Petersen and Cummins, 1974; Gazzera *et al.*, 1991; Salmoiraghi *et al.*, 1991).

Figure 3. The comparisons between land use categories (see methods) and the values of river quality bank indices, by ANOVA test with interaction line and error bars: ±1 Standard Deviation(s).

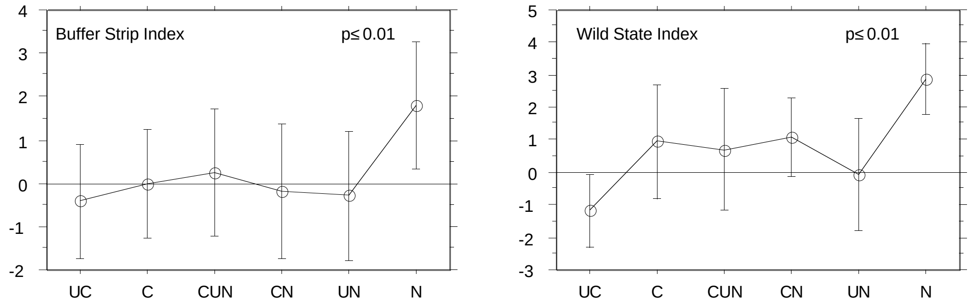
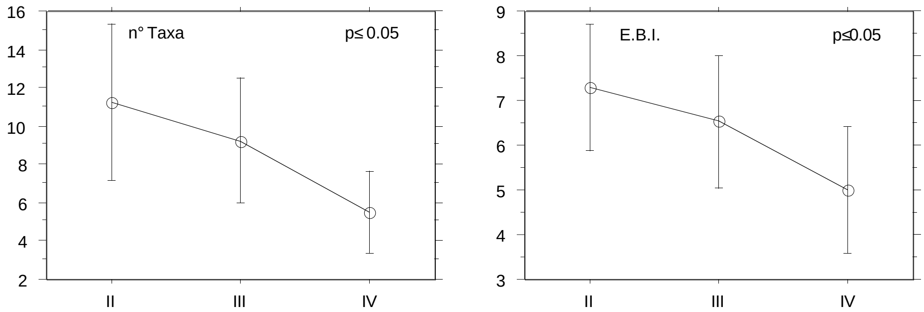
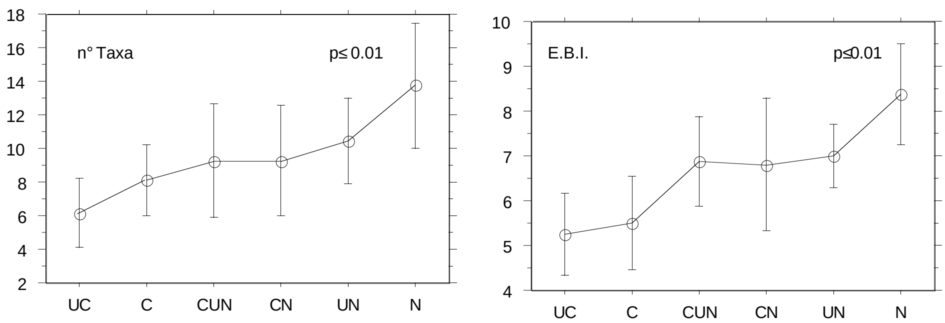


Figure 4. The comparisons between the quality classes of Wild State Index and macroinvertebrate biodiversity, by ANOVA test with interaction line and error bars: ±1 Standard Deviation(s).



The small streams are so closely linked to their catchments that terrestrial disturbances, such as agriculture and urbanisation, cause severe and long-term disruption which is likely to impact on both local and downstream environments. The study indicates that the land use near a riparian zone affects the structure of the zone which in turn modifies the structure of the aquatic communities. The ANOVA test shows highly significant ($p \leq 0.01$) differences between the different land uses, with respect to the macroinvertebrate biodiversity as measured by n° of taxa and E.B.I. (Fig. 5).

Figure 5. The comparisons between land use categories (see methods) and macroinvertebrate biodiversity, by ANOVA test with interaction line and error bars: ±1 Standard Deviation(s).



A river effect can be excluded because the different land uses considered are distributed over the different water courses (Table 3). A stronger land use by man coincides with the lowest EBI values and the lowest number of taxa.

Table 3. The sites divided by six different typologies of land use (see methods) for each water course of the Reno River Basin.

Water courses	*Land uses*					
	UC	*C*	*CUN*	*CN*	*UN*	*N*
Reno	2		3	3	3	3
Samoggia	1	1		1		1
Lavino	1		1	1		
Silla				1		1
Vergatello		1				
Venola			1			
Sambro						1
Limentra			1			1
Setta			1		1	3
Savena	1		1		1	2
Sellustra		5				
Senio	1	1	1	2		
Sinitria		1			1	1

CONCLUSIONS

The results of this study can be summarised by two points. The first is that the surrounding areas affect the riparian zone which, in turn, modifies the aquatic macroinvertebrate biodiversity because of the alterations of the riparian zone. The second is that the lotic macroinvertebrates are affected by the natural state of the riparian zone but not by its functions, such as filtering, metabolising and bioaccumulating nutrients and pollutants. Although further specific investigations are needed, it seems that the buffer zones in the sampling reaches are not “wide” enough to buffer the impact of terrestrial land use. We need to evaluate how “wide” the buffer zone should be and also understand better the role of this area in modifying the transport of nutrients, sediment and micropollutants to the stream (Mulholland, 1992).

In the last ten years the importance of buffer zones has been well-documented (Peterjohn and Correll, 1984; Haycock *et al.*, 1993; Brunet *et al.*, 1994; Lowrance *et al.*, 1995; Hubbard and Lowrance, 1996). Considering this extensive information we endeavoured to delineate the potential of the buffer zone to protect water quality. Unfortunately, the sites available did not allow us to do so, as there were few sites with the necessary combination of land over-use and efficient buffer zone. Consequently we were unable to evaluate the potential of the buffer zone in water protection in the Reno River Basin. Rehabilitation of riparian zones and their functions and clearly demonstrating the relationships involved is the first step in making this evaluation. Another task to be undertaken is to test the performance of the available buffer strip management techniques (Altier *et al.*, 1994), set up under different environmental conditions, in the study area.

As Petersen *et al.* (1987) explained, for economic, political and cultural reasons it is difficult to manage the entire drainage basin. Therefore, it is practical to start on a small scale, with the land areas of streamside vegetation, the riparian zones.

The rehabilitation of riparian zones would be improved by the application of EU regulation concerning the promotion of more environmentally friendly agricultural practices (Reg. CEE 2078/92) particularly the actions regarding 20-year set-aside and maintenance, restoration and conservation of natural areas.

The recently established “River Basin Authority” should be able to play a key role in river protection, overcoming the former sectorial approach where the river was seen simply as a conveyor of water with no environmental significance.

ACKNOWLEDGEMENTS

Special thanks to Lorenzo Canciani and Stefano Ramazza for their assistance.

The manuscript was refined through comments by Maria Giovanna Braioni, Gilles Pinay and Ugo Peruch.

REFERENCES

Altier, L.S., Lowrance, R.R., Williams, R.G., Sheridan, J.M., Bosch D.D., Hubbard, R.K., Mills, W.C. and Thomas, D.L. (1994) An ecosystem model for the management of riparian areas. In: *Riparian Ecosystem in the Humid U.S., Functions, Values and Management.* Wash., D.C.: Natl. Assoc. Conserv. Districts. pp. 373-387.

Anderson, N.H. and Cummins, K.W. (1979) Influences of diet on the life histories of aquatic insects. *J. Fish. Res. Bd. Can.*, **36**, 335-342.

Braioni, M.G., Braioni, A., De Franceschi, P., Mason, F., Ruffo, S., Sambugar, B., Modena, P., Ragusa, F. and Montresor, A. (1994) New environmental indices for assessing river bank quality: the case of the River Adige (North-East Italy). In: Link, G.L. and Naiman, R.J. (Eds.) *Proceedings of the International Workshop, the Ecology and Management of Aquatic-Terrestrial Ecotones* (Seattle 14-19 February 1994). Center for Streamside Studies, MAB, IHP University of Washington USA. pp 86-94.

Braioni, M.G. (in press) New Environmental indices for assessing bank quality in the restoration and the sustainable management of a river: the method. In: Zalewski, M., Thorpe, Y.E. and Schiemer, F. (Eds.) *Fish and Land Fisheries Science, Limnologie and Landscape Ecology.* M.A.B. Man and the Biosphere series vol.5. The Parthenon Publishing Group. pp: 1-40.

Brookes, A. (1988) Recovery and Restoration of Some Engineered British River Channels. In: Boon, P., Callow, P. and Petts, G. (Eds.) *River Conservation and Management.* John Wiley and Sons. pp 470.

Brunet, R.C., Pinay, G., Gazelle, F. and Roques, L. (1994) Role of the floodplain and riparian zone in suspended matter and nitrogen retention in the Adour River, south-west France. *Regulated Rivers; Research and Management*, **9**, 55-63.

Cattaneo, A., Salmoiraghi, G. and Gazzera, S. (1995) The Rivers of Italy. In: Cushing, C.E., Cummins, K.W. and Minshall, G.W. (Eds.) *River and Stream Ecosystems.* Ecosystems of the world 22. Elsevier. pp 479-505.

Cummins, K.W. and Klug, M.J. (1979) Feeding ecology of stream invertebrates. *Ann. Rev. Ecol. Syst.*, **10**, 147-172.

Cummins, K.W., Wilzbach, M.A., Gates, D.M., Perry, J.B. and Tagliaferro, W.B. (1989) Shredders and Riparian Vegetation. *Bioscence*, **39**(1), 24-30.

Fisher, S.G. (1977) Organic matter processing by a stream-segment ecosystem: Fort River, Massachusetts, U.S.A. *Internat. Revue der gesamten Hydrobiologie*, **62**, 701-727.

Gazzera, S., Cummins, K.W. and Salmoiraghi, G. (1991) A comparison of leaf litter processing in Maryland and Italian streams. *Verh. Int. Verein. Limnol.*, **24**, 1703-1706.

Ghetti, P.F. (1986) *I macroinvertebrati nell'analisi di qualità dei corsi d'acqua. Manuale di applicazione Indice Biotico: E.B.I. modificato.* Provincia Autonoma di Trento, pp 111.

Ghetti, P.F. and Bonazzi, G. (1981) *I macroinvertebrati nella sorveglianza ecologica dei corsi d'acqua.* Collana del progetto Finalizzato "Promozione della qualità dell'ambiente". C.N.R., AQ/1/127, 1-181.

Hubbard, R.K. and Lowrance, R.R. (1996) Solute transport and filtering through a riparian forest. *American Society of Agricultural Engineers.*, **39**(2), 477-488.

Haycock, N.E., Pinay, G. and Walker, C. (1993) Nitrogen retention in river corridors: European perspective. *Ambio*, **22**(6), 340-346.

Hellawell, J.M. (1986) *Biological indicators of fresh water pollution and environmental management.* Elsevier Applied Science Publish., London and New York.

Lowrance, R.R., Vellidis, G. and Hubbard, R.K. (1995) Denitrification in a restored riparian forest wetland. *J. Environ. Qual.*, **24**(5), 808-815.

Mulholland, P.J. (1992) Regulation of nutrient concentration in a temperate forest stream: Roles of upland, riparian and instream processes. *Limnol. Oceanogr.*, **37**(7), 1512-1526.

Naiman, R.J. and Décamps, H. (1990) *The Ecology and Management of Aquatic-Terrestrial Ecotones.* UNESCO, Paris, Man and the Biosphere Series, pp 316.

Peterjohn, W.T. and Correll, D.L. (1984) Nutrient dynamics in an agricultural watershed: Observations on the role of a riparian forest. *Ecology.*, **65**(5), 1466-1475.

Petersen, R.C. and Cummins, K.W. (1974) Leaf processing in a woodland stream. *Freshwater Biol.*, **4**, 343-368.

Petersen, R.C., Madsen, B.L., Wilzbach, M.A., Magadza, C.H.D., Paarlberg, A., Kullberg, A. and Cummins, K.W. (1987) Stream Management: Emerging Global Similarities. *Ambio*, **16**(4), 166-179.

Pinay, G., Décamps, H., Chauvet, E. and Fustec, E. (1990) Functions of ecotones in fluvial systems. In: Naiman, R.J. and Décamps, H. (Eds.) *The Ecology and Management of Aquatic-Terrestrial Ecotones.* UNESCO, Paris, Man and the Biosphere Series, pp 316.

Salmoiraghi, G., Nicolai, P. and Bigazzi, M. (1991) Aquatic insect emergence pattern in the Rio Torto. *Soc. Ital. Ecologia,* **12**, 697-702.

Rosemberg, D.M. and Resh, V.H. (1992) *Freshwater biomonitoring and benthic macroinvertebrates.* Chapman and Hall, London. pp 488.

Webster, J.R. and Benfield, E.F. (1986) Vascular plant breakdown in freshwater ecosystems. *Ann. Rev. Ecol. Syst.,* **17**, 567-594.

Woodiwiss, P.F. (1978) Compatibility study of biological-ecological assessment methods. *Second technical seminar on the River and tributaries.* Commission of the European Communities.

Contaminant effects on microbial functions in riparian buffer zones

10

P.M. Groffman

Institute of Ecosystem Studies, Box AB, Millbrook, NY 12545, USA.

Abstract

Micro-organisms are fundamental to several aspects of the pollutant removal capacity of buffer zones. Microbes can take up nitrate (NO_3^-) as a nutrient via immobilisation, or can convert it to nitrogen (N) gas via denitrification. Organic compounds, including animal wastes, pesticides and industrial compounds, can be degraded by a wide range of general and/or specific microbial groups. While much work has gone into determining the factors that control the nature and extent of microbial activity in soils and aquifers, much less work has gone into describing the effects of contaminants on soil microbes. Microbes are inherently controlled by the supply of energy (e.g. organic compounds or reduced inorganic compounds) and the availability of electron acceptors (e.g. oxygen (O_2), NO_3^-, sulphate (SO_4^{2-}), carbon dioxide). Contaminants can influence microbes either by providing energy (e.g. organic compounds), accessory nutrients (e.g. N, phosphorus), or electron acceptors (e.g. NO_3^-, SO_4^{2-}), or by affecting microbial growth and/or efficiency (e.g. heavy metals). In this paper, I review the ways that pollutants and microbes interact in soils and aquifers, in a general sense, and then discuss in detail the effects on microbes of the two classes of contaminants of most concern in riparian buffer zones (N and pesticides). I stress the need to design and interpret studies of microbial processes in field and landscape-scale contexts that are relevant to buffer zone research and management. Critical areas of uncertainty (e.g. N saturation) are highlighted.

INTRODUCTION

Although the role of micro-organisms in the attenuation of pollutants in riparian buffer zones is well-recognised and generally well-studied, the effect of contaminants on microbial processes is much less well-understood. Microbial processes that affect the dynamics of N and organic pollutants in buffer zones have received much attention over the last 20 years or so. However, the effects of pollutants on the microbial populations and communities that transform pollutants are not well-studied. These effects represent a critical gap in our understanding of the ability of riparian zones to function as long-term sinks for pollutants because of the possibility of positive and negative feedback effects of pollutants on microbial populations and processes.

In this paper, I give a brief review of the microbial processes responsible for pollutant removal in buffer zones. This review focuses on relationships between microbial processes and readily observable "map-able" and manageable features of buffer zones, i.e. it is an ecosystem and landscape-scale evaluation of riparian buffer zone microbiology. I then go on to discuss the effects of the two most common classes of contaminants of concern in riparian buffer zones (N and pesticides) on buffer zone microbial communities and the positive and negative feedback effects of these pollutants on buffer zone pollutant attenuation capacity.

SCALE-APPROPRIATE MICROBIAL ECOLOGY

Evaluation of the role of microbial processes in riparian buffer zone function is complicated by the fact that most microbial research is carried out at small, laboratory scales of investigation. Translating information from pure culture and molecular studies into an understanding of the fate and transport

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0 © 1997 Quest Environmental.

of pollutants in soils and groundwater is a major challenge in microbial ecology. This translation requires an understanding of how the factors that control the microbial process in question are expressed at different scales of investigation. However, scale-appropriate evaluation of controlling factors is a very useful approach to posing specific questions about microbial functions, allowing for evaluation of the unique characteristics of buffer zones that facilitate microbial attenuation of pollutants, management strategies that enhance attenuation and site-specific factors that influence pollutant fate.

To illustrate a scale-appropriate approach to microbial ecology, I use denitrification, the microbial process that converts NO_3^- to N gases (Table 1). This anaerobic process is one of the most valued and well-studied microbial processes in riparian buffer zones. Most denitrification research is carried out at the organism or molecular scale and is focused on the micro-scale factors that influence this process, particularly O_2 , NO_3^- and carbon (C). There has also been great interest in denitrification at the field scale, because this process can be a significant sink for fertiliser N applied to support crop growth. In field-scale studies, investigators have generally not focused on O_2, NO_3^- and C as controllers of denitrification, rather they have focused on the appropriate field-scale controllers of this process. Many studies have examined relationships between denitrification and soil moisture (a field-scale controller of O_2 availability to denitrifying micro-organisms), fertiliser input or nitrification rates (field-scale controllers of NO_3^- availability to denitrifiers) and residue dynamics or organic matter inputs (field-scale controllers of C availability to denitrifiers). At the landscape scale, research has focused on factors that influence the variation in soil water, NO_3^- supply and C supply among different fields (i.e. landscape components). Landscape-scale studies focus on variation in soil type (e.g. texture and natural drainage class) and plant community type (e.g. different natural ecosystem types, different agricultural management systems). Regional-scale studies focus on even broader factors that influence the distribution of soil and plant community types such as geomorphic features and human economic systems.

Table 1. Controlling factors of denitrification at different scales. Adapted from Groffman (1991).

Scale	*Controlling factors*
Organism	O_2, NO_3^-, C
Field	Soil H_2O, NO_3^- supply, C supply
Landscape	Soil type, plant community type
Regional	Geomorphology, land use
Global	Biome type, climate

For riparian buffer zones, most of our microbial studies need to be relevant at the field or landscape scale. Field-scale studies are relevant to analysis of specific microbial processes within a particular riparian buffer zone, e.g. how does seasonal variation in soil moisture influence denitrification or pesticide degradation. Landscape-scale studies are useful for evaluating and comparing microbial processes in riparian buffer zones on different soils, with different vegetation, e.g. how do denitrification and pesticide degradation vary in soils with different water-table levels or in grass versus forested buffer zones. The value of scale-appropriate microbial ecology is that it allows for evaluation of specific microbial processes in relation to factors used for planning, management and evaluation of buffer zones and thus facilitates interaction with scientists from other disciplines studying these areas (e.g. hydrologists, soil scientists) and with decision makers involved in management and policy efforts.

A scale-appropriate approach to microbial ecology will also be important for resolving many of the mechanistic and management uncertainties for buffer zones that are highlighted in the other chapters of this book. Many of these uncertainties arise from spatial and temporal discontinuities between hydrology (large and slow, especially in groundwater) and microbiology (small and fast). Studying and expressing microbial processes at scales relevant to the movement of water and pollutants in the landscape will be critical to resolving these problems.

MICROBE-POLLUTANT INTERACTIONS

Given a scale-appropriate approach to microbial ecology, what can we say about interactions between specific pollutants and microbial processes in riparian buffer zones? At the organism scale, contaminants affect micro-organisms either by providing energy, accessory nutrients or electron acceptors (compounds used in respiration), and/or by affecting microbial growth and efficiency. Pollutants ranging from pesticides to ammonium (NH_4^+) can be consumed, i.e. eaten by microbes as energy sources. Nutrient pollutants of concern in buffer zones (e.g. N and P) can be used as nutrients by microbes, just as they are used by terrestrial plants and algae. Several pollutants of concern (e.g. NO_3^-, SO_4^{2-}) can be respired as electron acceptors by microbes. This organism-scale framework of evaluating pollutants as potential energy sources, nutrients or electron acceptors for microbes provides a strong framework for predicting the fate of specific pollutants in buffer zones.

Although many studies have evaluated the behaviour of different pollutants in the organism-scale context described above, pollutant/microbe interactions in riparian buffer zones are complicated by long- versus short-term effects, by physical and chemical controls over biological processes and by feedback and interactions with the plant community. Physical, primarily hydrologic, processes that transport pollutants often control the nature and extent of pollutant interaction with buffer zone micro-organisms. Examples of hydrologic control include pollutants moving rapidly in surface runoff that have no chance of being processed by microbes, or pollutants moving in deep groundwater that do not interact with the organic-rich and biologically active surface zones of the soil. Chemical processes of importance include fixation and/or sorption of pollutants by clay and organic matter. Pollutant interactions with the plant community can be complex, e.g. nutrients can change the quality of plant material that is the food source for soil microbes. And finally, the organism-scale approach to evaluating the fate of pollutants is limited because it is based on a short-term, small-scale understanding of microbial ecology. Over time, microbial demands for energy, nutrients and electron acceptors can become saturated, or community composition can change, invalidating predictions about how a pollutant will behave in a particular buffer zone. Over space, many physical, chemical and biological processes interact making it difficult to predict microbial/pollutant interactions at the field and landscape scales necessary in buffer zone studies using an organism-scale approach.

In the sections below, I briefly review organism-scale pollutant/microbe interactions for N and pesticides and then discuss how specific pollutant/microbe interactions are regulated at field and landscape scales. I also discuss positive and negative feedback interactions between pollutants, microbes and plants that could influence the pollutant attenuation function of buffer zones in the long term.

Nitrogen

Many of the chapters of this book deal with N dynamics in buffer zones (e.g. Gilliam, this volume) and I do not present a comprehensive discussion of N dynamics here. Rather, I focus on the effects of N on microbial processes and highlight key areas where these effects can influence the N filtering performance of buffer zones.

Nitrogen can enter riparian buffer zones in three forms, NH_4^+, NO_3^- or organic N, and can be transported in surface runoff, groundwater or particulate matter (organic N only). At the organism scale, NH_4^+ can function as an energy source for one specific microbial group (nitrifying bacteria) and NO_3^- serves as an electron acceptor for a diverse group of facultatively anaerobic bacteria (e.g. denitrifiers). Ammonium and NO_3^- can also serve as sources of N to support general microbial and plant growth and activity. Nitrogen contained in organic compounds is processed as a by-product as a result of the use of the carbon in these compounds as an energy source by the general heterotrophic microbial community. As organic compounds are processed by this community, N is either released to (mineralised), or taken up from (immobilised), the environment, depending on the N content of the compound being degraded and the N demands of the active microbial community.

Although our organism-scale understanding of the interactions between different N forms and microbes provides a strong basis for predicting the effects of N on microbial processes in buffer zones, these effects are complicated by interactions with hydrology, seasonal variation and interactions with plant processes. As mentioned above, and discussed in other chapters in this book, hydrologic transport controls the nature and extent of N interactions with the buffer zone microbial community. The N compounds of environmental interest are all highly soluble and can be transported through buffer zones in surface runoff or beneath buffer zones in deep groundwater flow. These hydrologic bypass problems are exacerbated by the seasonal nature of biological processes. In temperate zones, much hydrologic transport of pollutants occurs during the biologically dormant season when plant and microbial processing of N are reduced. Evaluating N effects on microbial processes in buffer zones is also greatly complicated by interactions with plants. Plants have a strong appetite for N and they thus compete with microbes for this nutrient. Moreover, in the long term, plant litter quality and community composition are altered by N (Aber *et al.*, 1989). These changes affect the flow and availability of C to microbes, changing their activity in ways that are difficult to predict.

Critical uncertainties in our understanding of the effects of N on microbial processes in buffer zones that I discuss below include: 1) denitrification response to N loading, 2) subsurface dynamics, and 3) N saturation.

1. Denitrification response to N loading

It is well-established that buffer zones dominated by inherently wet, poorly-drained, wetland, hydric surface soils have a high capacity to consume NO_3^- via denitrification. These soils support the anaerobic conditions and high levels of organic matter necessary for denitrification (most denitrifiers are heterotrophs) (Cooper, 1990; Ambus and Lowrance, 1991; Schipper *et al.*, 1993; Groffman, 1994). Wet buffer zone soils have been reported to denitrify over 100 kg N ha^{-1} y^{-1} (Pinay *et al.*, 1993; Lowrance *et al.*, 1995; Groffman and Hanson, 1997).

It is also well-established that denitrification in wet soils responds positively to NO_3^- and C additions. Many wet soils have low rates of denitrification due to a lack of NO_3^- (Bowden, 1987). However, once exposed to NO_3^-, i.e. once they are used as buffer zones, denitrification rates increase rapidly (Broderick *et al.*, 1988; Warwick and Hill, 1988; Schipper *et al.*, 1991; Ambus and Christensen, 1993; Bengtsson and Bergwall, 1995). The ability of denitrification to respond rapidly to NO_3^- additions is important to efforts to manage and/or restore buffer zones. However, the upper limit of denitrification capacity that can be attained, and the factors that regulate the nature and extent of this capacity, have not been established.

Table 2. Denitrification response to NO_3^- additions to anaerobic soil cores taken from four different ecosystem types* on similar soils in Rhode Island, USA. Values are means of five soil cores taken from each plot in July, 1988. Values followed by different superscripts within a row are significantly different in a one-way analysis of variance with a Fisher's protected least significant difference test. Data from Groffman *et al.* (1991).

	Dry Forest	*Wet Forest*	*Tall fescue*	*Reed canary grass*
		g N ha^{-1} d^{-1}		
Control	1.1^{b}	13.1^{a}	1.0^{b}	1.0^{b}
NO_3^- added	1,306^{b}	1,402^{b}	17,208^{a}	15,208^{a}

* The dry forest was dominated by approximately 80-year-old *Quercus* trees and was on a well-drained soil. The wet forest was dominated by approximately 80-year-old *Acer rubrum* trees and was on a poorly-drained soil. The tall fescue *(Festuca arundinacea)* and reed canary grass *(Phalaris arundinacea)* plots were established on a well-drained soil and were two years old at the time of sampling. All soils were Inceptisols derived from glacio-fluvial deposits.

Groffman *et al.* (1991) compared denitrification response to nitrate additions in four ecosystem types located on similar soil types. Response varied greatly, and in surprising ways, with two grass ecosystems showing much greater response than two forest ecosystems (Table 2). These differences suggest that the nature and extent of denitrification response to N loading depends fundamentally on organic matter quality at a site. The grasses are likely to have higher litter quality than the forests due to their inherent chemistry and to the use of fertiliser and lime for their production. It is not clear if denitrification response in the forests would eventually equal that in the grasses. However, these results strongly suggest that denitrification response to N loading is variable and amenable to management.

2. Subsurface N dynamics

The nature and extent of microbial activity in groundwater and aquifer material are highly uncertain. Subsurface environments are difficult to sample and studies have produced conflicting results, i.e. studies have found high potential and/or actual denitrification in subsurface material (Trudell *et al.*, 1986; Slater and Capone, 1987; Smith and Duff, 1988; Francis *et al.*, 1989; Obenhuber and Lowrance, 1991; Haycock and Pinay, 1993), while others have found little or no activity (Parkin and Meisinger, 1989; Groffman *et al.*, 1992; Bradley *et al.*, 1992; Lowrance, 1992; Yeomans *et al.*, 1992; Starr and Gillham, 1993; Groffman *et al.*, 1996). Subsurface environments are often very low in C, limiting the potential for heterotrophic activity which, in turn, regulates O_2 status (i.e. heterotrophic activity consumes O_2) (Lind and Eiland, 1989; Hiscock *et al.*, 1991; Johnson and Wood, 1992; Korom, 1992; McCarty and Bremner, 1992; Starr and Gillham, 1993; Weier and MacRae, 1993; Spalding and Parrott, 1994; Desimone and Howes, 1996). There is a critical need for studies to determine the field and landscape-scale factors controlling: 1) the inherent C content of subsurface materials (e.g. buried river channels, Fustec *et al.*, 1991; Haycock and Burt, 1993), and 2) C transport from the surface to the subsurface (e.g. root dynamics or dissolved organic C leaching). The importance of non-C based energy sources (e.g. sulphur, iron, methane) is also deserving of investigation (Pedersen *et al.*, 1991; Postma *et al.*, 1991; Korom, 1992; Garcia-Gil and Golterman, 1993; Parkin and Simpkins, 1993). The uncertainty surrounding microbial activity in the subsurface is important because in many areas, the dominant vector of NO_3^- transport from uplands into buffer zones is via groundwater (Hill, 1990; Jordan, *et al.*, 1993; Schnabel *et al.*, 1994).

Table 3. Microbial and root response to 10 months of NO_3^- dosing of aquifer sediments in a riparian buffer zone in Rhode Island, USA. Values for microbial variables are mean (standard error) of three replicate samples taken from two aquifer depths beneath three different soils within the riparian zone at four sampling dates between March 1992 and February 1993 (n=60). Values for roots are from only one depth (n=30).

	Control	*Dosed*
Denitrification enzyme activity ($\mu g\ N\ kg^{-1}\ h^{-1}$)	ND	ND
Microbial biomass C ($mg\ C\ kg^{-1}$)	62 (1)	67 (12)
Microbial biomass N ($mg\ N\ kg^{-1}$)	2.9 (0.3)	2.8 (0.2)
Root biomass ($mg\ kg^{-1}$)	203 (35)	129 (18)

ND – not detectable

Our lack of basic understanding of the nature and extent of microbial processes in the subsurface makes it very difficult to predict the effects on N additions on these processes. Groffman *et al.* (1996) quantified microbial response to 10 months of NO_3^- dosing of aquifer sediments. Surprisingly, there was no increase in denitrification potential, microbial biomass C content or microbial biomass N content in this study, highlighting the uncertainty over the factors regulating microbial biomass and activity in the subsurface (Table 3). It is possible that the lack of response to N additions was related to the significant decrease in root biomass in the dosed sediments which may have created strong C limitation of the microbial community.

3. N saturation

The long-term effects of N loading of buffer zones on microbial processes are not well-characterised. While denitrification results in clear cut removal of N from a buffer zone, plant uptake and immobilisation of N allow for recycling and possible future mineralisation, nitrification and loss of N. There is concern that long-term loading of plant and microbial pools will result in N saturation of buffer zones, a condition where N outputs equal N inputs, i.e. the buffer zone will no longer be a sink for N (Aber *et al.*, 1989).

Hanson *et al.* (1994a,b) observed enrichment of total plant and microbial N pools and marked increases in soil NO_3^- pools, mineralisation and nitrification (clear symptoms of N saturation) in a forested riparian buffer zone subjected to long-term (30 year) N loading relative to a control riparian zone (Figure 1). The N loading essentially converted an N-poor site into an N-rich site, with an inherently lower ability to retain N inputs and a higher susceptibility to high rates of N loss following disturbance by cutting, fire, blowdown, etc. (Vitousek *et al.*, 1982). Interestingly, this site still functioned as an effective buffer zone for N, likely due to the strong denitrification response to the N loading.

There is a clear need for studies to partition the fate of N that enters buffer zones between denitrification and internal recycling and to trace the ultimate fate of N immobilised by plants and microbes. This partitioning and fate should be controlled by water-table dynamics and soil and vegetation types, i.e. field and landscape-scale variables amenable to practical assessment and management. There are also concerns that N enrichment of buffer zones can lead to changes in plant communities (Morris, 1991; Ehrenfeld and Schneider, 1991) which can support different rates of N mineralisation (Hill and Shackleton, 1989; van Vuuren *et al.*, 1992) and biodiversity (Naiman *et al.*, 1993).

Pesticides

Although the fate and transport of pesticides in buffer zones is important in agricultural, urban/suburban and forestry contexts, there have been very few studies that have addressed pesticide dynamics in buffer zones. Pesticide/microbial interactions are very well-studied from two viewpoints: 1) the effects of pesticides on microbial processes such as respiration, N mineralisation, nitrification and denitrification, and 2) the ability of microbes to degrade specific pesticides. The first group of studies has found that the vast majority of pesticides have little or no effect on microbial processes in soil (Goring and Laskowski, 1982; Grant and Payne, 1982; Domsch, 1984; Bradley *et al.*, 1994; Martens and Bremner, 1994; Ghani *et al.*, 1996). The biggest exception to this result are fungicides, which can eliminate a significant fraction of the soil microbial biomass (Anderson *et al.*, 1981; Duah-Yentumi and Johnson, 1986). The second group of studies has found that microbial populations capable of degrading many pesticides can develop quite rapidly following repeated application of the pesticides (Spain and van Veld, 1983; Roeth, 1986; Smith and Aubin, 1991). Unfortunately, there have been few field or landscape-scale studies of pesticide/microbial interactions relevant to buffer zones.

The few studies that have investigated pesticide/microbial interactions in riparian buffer zones have suggested that these interactions are complex and worthy of further study. Entry *et al.* (1994, 1995) found that herbicide degradation was faster in forest than pasture riparian zones and that degradation was faster in old-growth forests than in younger forests. In contrast, Paterson and Schnoor (1992) found no difference in herbicide loss from, or biotransformation within, barren, maize-cropped or forested riparian zones. Voos and Groffman (in press, a) found that herbicide degradation was faster in forested soils than in grass or maize-cropped soils in a laboratory study (Table 4). However, in a companion field study (Voos and Groffman in press, b), physico-chemical sorption in the highly organic surface soil horizons of the forest soils inhibited microbial access to, and degradation of, herbicides in the forest soils relative to the grass or maize-cropped soils.

Figure 1. Soil NO_3^-, potential net nitrification and mineralisation, and denitrification rate in NO_3^- enriched and control riparian forests in Rhode Island, USA. Values are means (standard error) over 15 sampling dates between March 1991 and 1992 in four soil drainage classes at each site. MWD = moderately well drained, SPD – somewhat poorly drained, VPD = very poorly drained. Data from Hanson *et al.* (1994a, b).

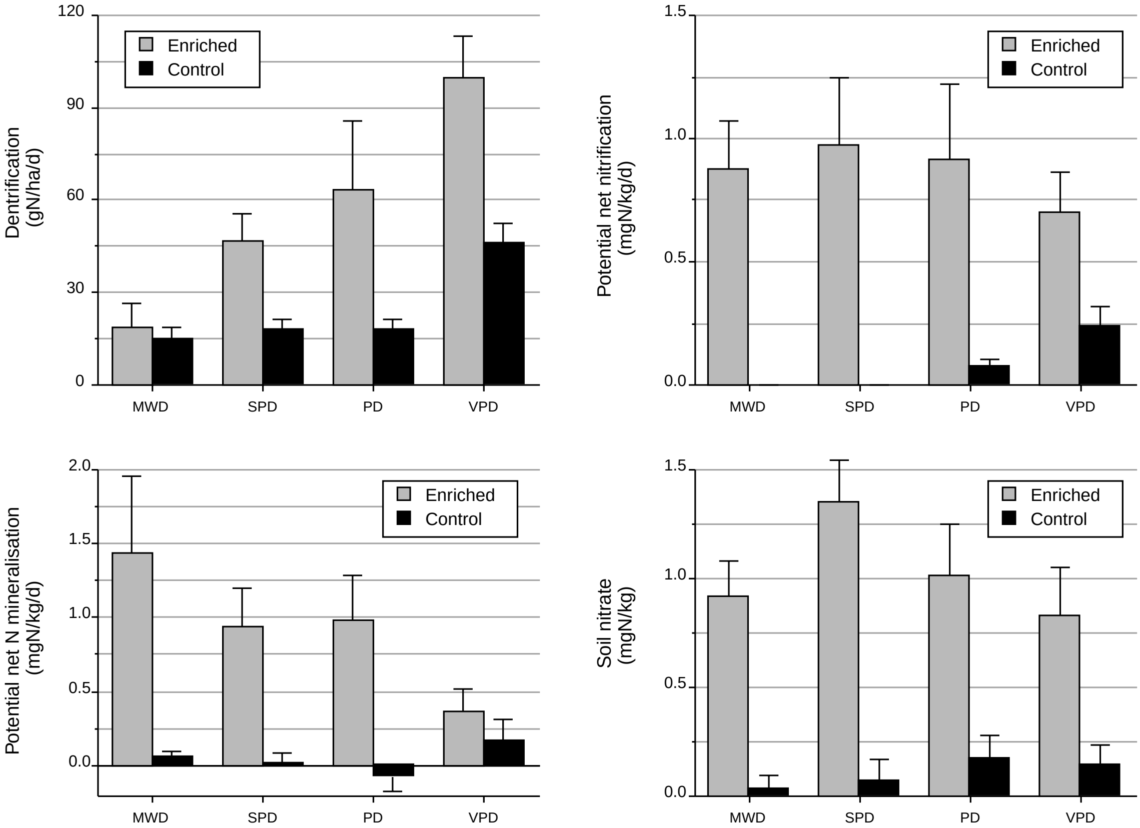


Table 4. Residual herbicide (dicamba) concentrations in four different ecosystem types* on similar soils in Rhode Island, USA. Values are means of three samples taken from laboratory microcosms or field plots after 80 days of incubation. Values followed by different superscripts within a row are significantly different in a one-way analysis of variance with a Fisher's protected least significant difference test. Laboratory data from Voos and Groffman (in press, a), field data from Voos and Groffman (in press, b).

	Laboratory study	*Field study*
	mg kg^{-1}	
Corn	1.5 b	0.0b
Sod	2.1ab	0.0b
Dry forest	0.0c	0.4a
Wet forest	0.0 c	0.9a
Aquifer	2.6a	0.0b

* The dry forest was dominated by approximately 80-year-old *Quercus* trees and was on a well-drained soil. The wet forest was dominated by approximately 80-year-old *Acer rubrum* trees and was on a poorly-drained soil. The corn *(Zea mays)* and sod (primarily *Poa pretensis*) plots were established on a well-drained soil and had been under current management for approximately eight years at the time of sampling. The aquifer material was collected 60 cm below the water table and 160 cm below the soil surface from a previously-excavated site within the dry forest. All soils were Inceptisols derived from glacio-fluvial deposits.

Similar to N, there is great uncertainty about the potential for degradation of pesticides in groundwater in buffer zones. Voos and Groffman (in press, a) observed very slow rates of herbicide degradation in riparian forest aquifer material relative to forest, maize-cropped or grass surface soils (Table 4). Beare *et al.* (1994) observed very slow rates of aldicarb degradation in groundwater material from a forested buffer zone. The fate of pesticides in groundwater is an active area of research. However, most pesticide in groundwater studies are done in upland areas where groundwater is deep below the biologically active zone of the soil. In buffer zones, groundwater is often closer to the surface, moving through sediments with higher C and lower O2 than upland areas.

CONCLUSIONS AND RECOMMENDATIONS

While there is high confidence in the ability of denitrification to consume large amounts of NO_3^- in surface soils, there is a strong need for research to determine how management and restoration activities can maximise denitrification capacity, e.g. different plants, harvesting regimes, soil liming.

There is a strong need for research on N saturation in buffer zones. We need to determine if it is possible to manipulate the partitioning of N that moves into buffer zones between plants and denitrification towards more denitrification. If this partitioning cannot be manipulated, there is a strong need to develop management strategies to remove N from buffer zones (e.g. via harvesting).

There is a strong need to develop better understanding and management of groundwater microbiology. We need to determine if surface management (e.g. vegetation selection) influences subsurface microbial activity and to develop a capacity to predict where and when groundwater denitrification will occur.

Existing research on pesticide/microbial interactions in buffer zones is quite limited and should be expanded. Data suggest that most pesticides will not affect microbial nutrient cycling (e.g. N mineralisation) and water quality maintenance (e.g. denitrification) functions in buffer zones. However, the capacity of buffer zone microbial communities to develop the capacity to degrade specific pesticides should be investigated. The importance of physical factors (e.g. sorption) as regulators of pesticide/microbial interactions in different types of buffer zones should also be assessed.

REFERENCES

Aber, J.D., Nadelhoffer, K.J., Steudler, P. and Melillo, J.M. (1989) Nitrogen saturation in northern forest ecosystems. *BioScience,* **39**, 378-386.

Ambus, P. and Christensen, S. (1993) Denitrification variability and control in a riparian fen irrigated with agricultural drainage water. *Soil Biol. Biochem.*, **25**, 915-923.

Ambus, P. and Lowrance, R. (1991) Comparison of denitrification in two riparian soils. *Soil Sci. Soc. Am. J.*, **55**, 994-997.

Anderson, J.P.E., Armstrong, R.A. and Smith, S.N. (1981) Methods to evaluate pesticide damage to the biomass of the soil microflora. *Soil Biol. Biochem.,* **13**, 149-153.

Beare, M.H., Lowrance, R.R. and Meyer, J.L. (1994) Biotic regulation of NO_3^- depletion in a coastal plain riparian forest: Experimental approach and preliminary results. In: *Riparian Ecosystems in the Humid US.* National Association of Conservation Districts, Washington, D.C. pp 388-397.

Bengtsson, G. and Bergwall, C. (1995) Heterotrophic denitrification potential as an adaptive response in groundwater bacteria. *FEMS Microbiol. Ecol.*, **16**, 307-318.

Bowden, W.B. (1987) The biogeochemistry of nitrogen in freshwater wetlands. *Biogeochemistry*, **4**, 313-348.

Bradley, P.M., Chapelle, F.H., Jagucki, M.L. and McMahon, P.B. (1994) Effect of atrazine on potential denitrification in aquifer sediments. *Soil Biol. Biochem.,* **26**, 523-524.

Bradley, P.M., Fernandez Jr., M. and Chapelle, F.H. (1992) Carbon limitation of denitrification in an anaerobic groundwater system. *Environ. Sci. Technol.*, **12**, 2377-2381.

Broderick, S.J., Cullen, P. and Maher, W. (1988) Denitrification in a natural wetland receiving secondary treated effluent. *Wat. Res.*, **4**, 431-439.

Cooper, A.B. (1990) Nitrate depletion in the riparian zone and stream channel of a small headwater catchment. *Hydrobiol.*, **202**, 13-26.

Desimone, L.A. and Howes, B.L. (1996) Denitrification and nitrogen transport in a coastal aquifer receiving wastewater discharge. *Environ. Sci. Technol.*, **30**, 1152-1162.

Domsch, K.H. (1984) Effects of pesticides and heavy metals on biological processes in soil. *Plant and Soil*, **6**, 367-378.

Duah-Yentumi, S. and Johnson, D.B. (1986) Changes in soil microflora in response to repeated applications of some pesticides. *Soil Biol. Biochem.*, **18**, 629-635.

Ehrenfeld, J.G. and Schneider, J.P. (1991) Chamaecyparis thyoides wetlands and suburbanisation: Effects on hydrology, water quality and plant community composition. *J. Appl. Ecol.*, **28**, 467-490.

Entry, J.A., Donelly, P.K. and Emmingham, W.H. (1995) Atrazine and 2,4-D mineralisation in relation to microbial biomass in soils of young-, second- and old-growth riparian forests. *Appl. Soil Ecol.*, **2**, 77-84.

Entry, J.A., Donelly, P.K. and Emmingham, W.H. (1994) Microbial mineralisation of atrazine and 2,4-dichlorophenoxyacetic acid in riparian pasture and forest soils. *Biol. Fert. Soils*, **18**, 89-94

Francis, A.J., Slater, J.M. and Dodge, C.J. (1989) Denitrification in deep subsurface sediments. *Geomicrobiol. J.*, **7**, 103-116.

Fustec, E., Mariotti, A., Grillo, X. and Sajus, J. (1991) Nitrate removal by denitrification in alluvial ground water: Role of a former channel. *J. Hydrol.*, **123**, 337-354.

Garcia-Gil, L.J. and Golterman, H.L. (1993) Kinetics of FeS-mediated denitrification in sediments from the Camargue (Rhone delta, southern France). *FEMS Microbiol. Ecol.*, **13**, 85-92.

Ghani, A., Wardle, D.A., Rahman, A. and Lauren, D.R. (1996) Interactions between 14C-labelled atrazine and the soil microbial biomass in relation to herbicide degradation. *Biol. Fert. Soils*, **21**, 17-22.

Goring, C.A.I. and Laskowski, D.A. (1982) The effects of pesticides on nitrogen transformations in soils. In: Stevenson, F.J. (Ed.) *Nitrogen in Agricultural Soils.* American Society of Agronomy, Madison, WI. pp 689-720.

Grant, M.A. and Payne, W.J. (1982) Effects of pesticides on denitrifying activity in salt marsh sediments. *J. Environ. Qual.*, **11**, 369-371.

Groffman, P.M. (1994) Denitrification in freshwater wetlands. *Current Topics in Wetland Biogeochemistry*, **1**, 15-35.

Groffman, P.M., Axelrod, E.A. Lemunyon, J.L. and Sullivan, W.M. (1991) Denitrification in grass and forest vegetated filter strips. *J. Environ. Qual.*, **20**, 671-674.

Groffman, P.M., Gold A.J. and Simmons, R.C. (1992) Nitrate dynamics in riparian forests: Microbial studies. *J. Environ. Qual.*, **21**, 666-671.

Groffman, P.M. and Hanson, G.C. (in press). Denitrification in red maple wetlands. *Soil Sci. Soc. Am. J. 1997.*

Groffman, P.M., Howard, G., Gold A.J. and Nelson, W.M. (in press) Microbial nitrate processing in shallow groundwater in a riparian forest. *J. Environ. Qual. 1966.*

Hanson, G.C., Groffman, P.M. and Gold, A.J. (1994a) Denitrification in riparian wetlands receiving high and low groundwater nitrate inputs. *J. Environ. Qual.*, **23**, 917-922.

Hanson, G.C., Groffman, P.M. and Gold, A.J. (1994b) Symptoms of nitrogen saturation in a riparian wetland. *Ecol. Appl.*, **4**, 750-756.

Haycock, N.E. and Burt, T.P. (1993) Role of floodplain sediments in reducing the nitrate concentration of subsurface run-off: A case study in the Cotswolds, UK. *Hydrological Processes*, **7**, 287-295.

Haycock, N.E. and Pinay, G. (1993) Groundwater nitrate dynamics in grass and poplar vegetated riparian buffer strips during the winter. *J. Environ. Qual.*, **22**, 273-278.

Hill, A.R. (1990) Ground water flow paths in relation to nitrogen chemistry in the near-stream zone. *Hydobiol.*, **206**, 39-52.

Hill, A.R. and Shackleton, M. (1989) Soil N mineralisation and nitrification in relation to nitrogen solution chemistry in a small forested watershed. *Biogeochem.*, **8**, 167-184.

Hiscock, K.M., Lloyd, J.W. and Lerner, D.N. (1991) Review of natural and artificial denitrification of groundwater. *Wat. Res.*, **9**, 1099-1111.

Johnson, A.C. and Wood, M. (1992) Microbial potential of sandy aquifer material in the London basin. *Geomicrobiol. J.*, **10**, 1-13.

Jordan, T.E., Correll, D.L. and Weller, D.E. (1993) Nutrient interception by a riparian forest receiving inputs from adjacent cropland. *J. Environ. Qual.*, **22**, 467-473.

Korom, S.F. (1992) Natural denitrification in the saturated zone: A review. *Wat. Resour. Res.*, **28**, 1657-1668.

Lind, A.-M. and Eiland, F. (1989) Microbiological characterisation and nitrate reduction in subsurface soils. *Biol. Fert. Soils*, **8**, 197-203.

Lowrance, R. (1992) Groundwater nitrate and denitrification in a coastal plain riparian forest. *J. Environ. Qual.*, **21**, 401-405.

Lowrance, R.R., Todd, R.L. and Asmussen, L.E. (1984) Nutrient cycling in an agricultural watershed: I. Phreatic movement. *J. Environ. Qual.*, **13**, 22-27.

Lowrance, R., Vellidis, G. and Hubbard, R.K. (1995) Denitrification in a restored riparian forest wetland. *J. Environ. Qual.*, **24**, 808-815.

Martens, D.A. and Bremner, J.M. (1994) Effects of preemergence and postemergence herbicides on urea hydrolysis and nitrification of urea nitrogen in soil. *Biol. Fert. Soils*, **17**, 309-313.

McCarty, G.W. and Bremner, J.M. (1992) Availability of organic carbon for denitrification of nitrate in subsoils. *Biol. Fert. Soils*, **14**, 219-22.

Morris, J.T. (1991) Effects of nitrogen loading on wetland ecosystems with particular reference to atmospheric deposition. *Ann. Rev. Ecol. Syst.*, **22**, 257-279.
Naiman, R.J., Decamps, H. and Pollock, M. (1993) The role of riparian corridors in maintaining regional biodiversity. *Ecol. Appl.*, **3**, 209-212.
Obenhuber, D.C. and Lowrance, R. (1991) Reduction of nitrate in aquifer microcosms by carbon additions. *J. Environ. Qual.*, **20**, 255-258.
Parkin, T.B. and Meisinger, J.J. (1989) Denitrification below the crop rooting zone as influenced by surface tillage. *J. Environ. Qual.*, **18**, 12-16.
Parkin, T.B. and Simpkins, W.W. (1993) Methane as an energy source for denitrification in groundwater. *Agron. Abstr.*, **85**, 256.
Paterson, K.G. and Schnoor, J.L. (1992) Fate of alachlor and atrazine in a riparian zone field site. *Wat. Environ. Res.*, **64**, 274-283.
Pedersen, J.K., Bjerg, P.L. and Christensen, T.H. (1991) Correlation of nitrate profiles with groundwater and sediment characteristics in a shallow sandy aquifer. *J. Hydrol.*, **124**, 263-277.
Pinay, G., Roques, L. and Fabre, A. (1993) Spatial and temporal patterns of denitrification in a riparian forest. *J. Appl. Ecol.*, **30**, 581-591.
Postma, D., Boesen, C., Kristiansen, H. and Larsen, F. (1991) Nitrate reduction in an unconfined sandy aquifer: Water chemistry, reduction processes and geochemical modeling. *Wat. Resour. Res.*, **27**, 2027-2045.
Roeth, F.W. (1986) Enhanced herbicide degradation in soil with repeat application. *Rev. Weed Sci.*, **2**, 45-65.
Schipper, L.A., Cooper, A.B., Harfoot, C.G. and Dyck, W.J. (1993) Regulators of denitrification in an organic riparian soil. *Soil Biol. Biochem.*, **25**, 925-933.
Schipper, L.A., Cooper, A.B. and Dyck, W.J. (1991) Mitigating nonpoint-source nitrate pollution by riparian-zone denitrification. In: Bogardi, I., Kuzelka, R.D. and Ennenga, W.G. (Eds.) *Nitrate Contamination: Exposure, Consequences, and Control.* Springer-Verlag, Berlin Heidelberg. pp 401-413.
Schnabel, R.R., Gburek, W.J. and Stout, W.L. (1994) Evaluating riparian zone control on nitrogen entry into Northeast Streams. In: *Riparian Ecosystems in the Humid US.* National Association of Conservation Districts, Washington, D.C. pp 432-445.
Slater, J.M. and Capone, D.G. (1987) Denitrification in aquifer soil and nearshore marine sediments influenced by groundwater nitrate. *Appl. Environ. Microbiol.*, **53**, 1292-1297.
Smith, A.E. and Aubin, A.J. (1991) Effects of long-term 2,4-D and MCPA field applications on the soil breakdown of 2,4-D, MCPA, mecoprop and 2,4,5-T. *J. Environ. Qual.*, **20**, 436-438
Smith, R.L. and Duff, J.H. (1988) Denitrification in a sand and gravel aquifer. *Appl. Environ. Microbiol.*, **54**, 1071-1978.
Spain, J.C. and Van Veld, P.A. (1983) Adaptation of natural microbial communities to degradation of xenobiotic compounds: Effects of concentration, exposure time, inoculum and chemical structure. *Appl. Environ. Microbiol.*, **45**, 428-535.
Spalding, R.F. and Parrott, J.D. (1994) Shallow groundwater denitrification. *Sci. Total Environ.*, **141**, 17-25.
Starr, R.C. and Gillham, R.W. (1993) Denitrification and organic carbon availability in two aquifers. *Ground Wat.*, **31**, 934-947.
Trudell, M.R., Gillham, R.W. and Cherry, J.A. (1986) An *in situ* study of the occurrence and rate of denitrification in a shallow unconfined sand aquifer. *J. Hydrol.*, **83**, 251-268.
van Vuuren, M.M.I., Aerts, R., Berendse, F. and De Visser, W. (1992) Nitrogen mineralisation in heathland ecosystems dominated by different plant species. *Biogeochem.*, **16**, 151-166.
Vitousek, P.M., Gosz, J.R., Grier, C.C., Melillo, J.M., Reiners, W.A. and Todd, R.L. (1982) A comparative analysis of potential nitrification and nitrate mobility in forest ecosystems. *Ecol. Monogr.*, **52**, 155-177.
Voos, G.V. and Groffman, P.M. (in press, a) Relationships between microbial biomass and herbicide degradation capacity in soils. *Biol. Fert. Soils.*
Voos, G.V. and P.M. Groffman. (in press, b) Herbicide dissipation in a heterogeneous landscape. *Appl. Soil Ecol.*
Warwick, J. and Hill, A.R. (1988) Nitrate depletion in the riparian zone of a small woodland stream. *Hydrobiol.*, **157**, 231-240.
Weier, K.L. and MacRae, I.C. (1993) Net mineralisation, net nitrification and potentially available nitrogen in the subsoil beneath a cultivated crop and a permanent pasture. *J. Soil Sci.*, **44**, 451-458.
Yeomans, J.C., Bremner, J.M. and McCarty, G.W. (1992) Denitrification capacity and denitrification potential of subsurface soils. Commun. *Soil Sci. Plant Anal.*, **23**, 919-927.

The influence of sedimentation on vegetation structure

11

J. Girel[1,2] and G. Pautou[1]

[1] UPR-ES 592 Laboratoire Ecosystèmes Alpins, Groupe Hydrosystèmes Alpins, Université Joseph Fourier, Centre de Biologie Alpine, B.P. 53, 38041 Grenoble (cedex 9), France
[2] ESA C.N.R.S. 5023 (Ecologie des Eaux douces et des Grands Fleuves)

Abstract

The upper Rhône river system enables us to highlight two types of major processes occurring along the river. These are (i) the trapping processes of sediments which initiated the deposition patterns and induced impacts on vegetation and (ii), the role of the plant communities on the hydrosystem functioning. Two major sedimentation patterns were studied:

1) The sedimentation pattern which characterises the rivers having a large floodplain. Here the water flow may move laterally and induce the establishment of bars, islands and terraces colonised by plant communities which are characteristic of various successional stages.
2) The sedimentation pattern which characterises the stream corridors disturbed by dyking and channelisation works since the end of the 18th century.

The latter case study was interesting for two reasons: (i) it allows us to analyse the sedimentation processes and plant colonisation that take place on the side bars within the dyked narrow floodplain and (ii) it provides information concerning the artificial sedimentation processes induced by engineering-works during the 19th century in order to make farming soils. The composition of plant communities that grew and succeeded while the fine sediment layer has been raised and the evolution over time of the warping basins provide interesting data. This information is useful before the development of "polder" and other various techniques planned to stock water and sediment flows within the valleys in order to prevent the flooding of agricultural and urban areas and to conserve natural habitats and biodiversity. In addition, hydroelectric developments have at times modified the dispersion patterns of water-, energy- and matter-fluxes, sometimes for several decades. Consequently, new conditions concerning erosion, sedimentation, hydrology and plant succession patterns are analysed.

INTRODUCTION: SITES FAVOURING THE SEDIMENTATION PROCESS

Stream corridors of the Alpine piedmont, through which water, energy, matter and species have travelled for centuries, have the potential to contribute towards a greater understanding of the complex interrelationships between sedimentation and vegetation. Some useful concepts are first recalled, followed by a description of the processes governing the alluvial system. The role of the factors controlling both sedimentation and plant succession are then discussed. Case studies are taken from the French Upper Rhône river system with particular emphasis on the interactions of riparian vegetation on sedimentation processes over various time scales. A comparative study is undertaken of a relatively undisturbed floodplain and a dyked stream with its abandoned floodplain.

Several authors (Morizawa, 1985; Kellerhals and Miles, 1996) describe five river channel patterns which can be divided into two main groups: (i) straight, sinuous or meandering river channels composed of a single main channel, (ii) rivers with multiple channels taking either the braided forms (with unstable bars and channels) or the form of an anastomosed channel (with more stable bars, mid-channel islands and secondary branches). Both slope and total discharge control velocity, energy

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0

fluxes and the transport of sediment, allowing for 4 distinct zones. From its source in the alpine headwaters to the mouth of an ocean or lake these zones are (Junk and Welcomme, 1990; Brookes, 1995):

1 - Upland areas, constrained by valley walls, with a steep gradient and coarse sediments (boulders and cobbles)
2 - Piedmont areas, with a moderate gradient and coarse sediments (cobbles, gravel and sand)
3 - Lowland areas, with a low gradient and fine sediments (fine sand, silt and clay)
4 - Estuarine areas (oceans, lakes), influencing tidal processes have low gradients and fine deposits.

Upland areas are characterised by 1st and 2nd order streams. The streamside vegetation comprises a narrow riparian fringe. These zones supply reaches further downstream with sediment. The piedmont zone is characterised by multiple braided or anastomosed channels with large numbers of islets and deposits of alluvium. Braided and anastomosed forms are found where the banks are erodible, the gradient steep and the bedload abundant. The braided form is found where, apart from during floods, the flow is insufficient to entrain the bedload; the result is the creation of numerous unstable alluvial deposits (Murray and Paola, 1994). Anastomosed streams are associated with sediments which remain in basins with a shallow gradient, or where there is a large enough supply of fine materials to maintain a broad floodplain; they are frequently found in zones of confluence where local base levels are rising (Petts and Foster, 1985). The banks and the islets become vegetated quickly, so landforms are generally more stable. Downstream, sections (of order 5 or above) have low gradients and the load is mainly fine material. These sections are characterised by erosion on the outside banks and deposition on the inside banks of bends to form point bars. The result of this is the sinuous or meandering form characterised by one principal channel with features such as mid-channel islets, cut-offs and oxbow lakes. Braided, anastomosed and meandering forms are made up transport and storage areas (2 and 3, see Fig. 1) which develop along Alpine rivers such as the Rhône river upstream of Lyon and along the Isère river between Albertville and Grenoble (Fig. 2). The two major types of aggradation promoting sedimentation are (i) deposition on point bars along inside bends of the streams and (ii) deposition during periods of high flow and flooding.

Figure 1. Geomorphic zones and floodplain models.

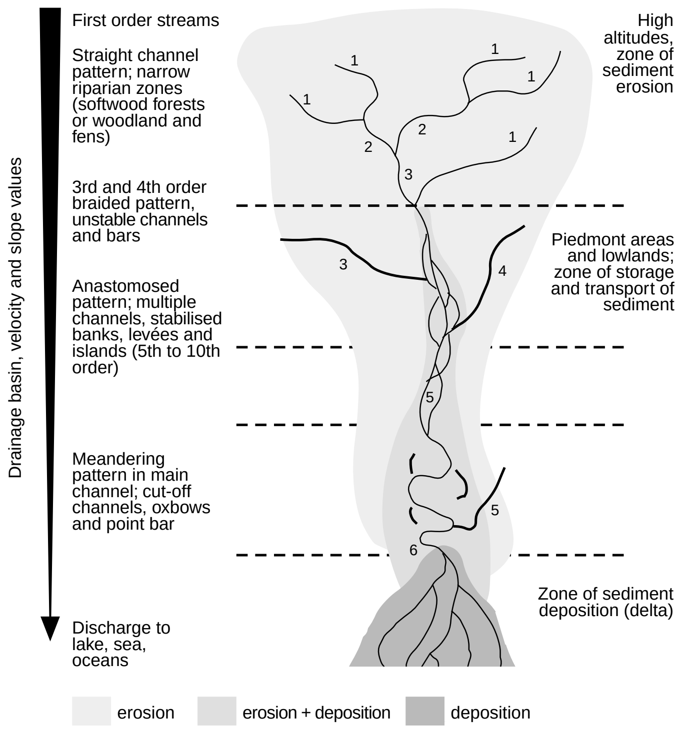
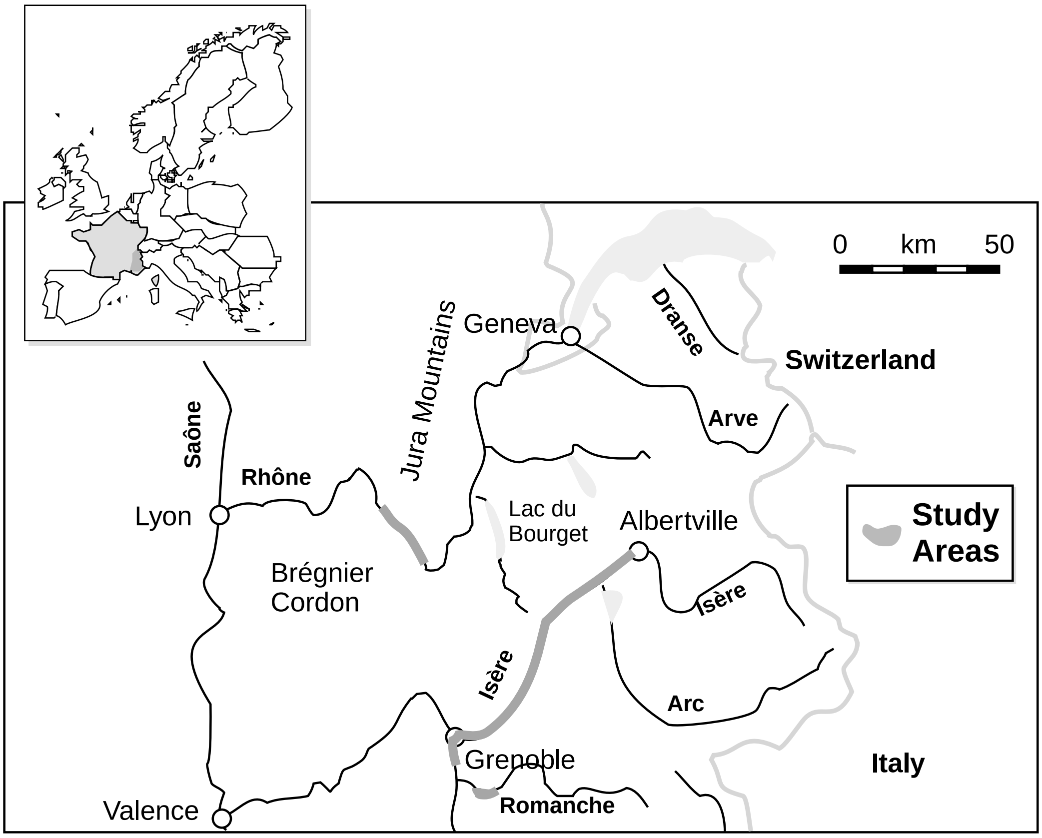


Figure 2. The French Upper Rhône River and major tributaries.



EROSION, TRANSPORT AND DEPOSITION OF SEDIMENT ON FLOODPLAINS

The amount of sediment transported to and by streams depends on basin area and shape in addition to steepness, vegetation coverage and land use. The amount of material flowing at any one time is a function of prevailing climatic factors. Once in the stream, the particles move in "pulses" during periods of high flow (Forman, 1995). Water flow controls or limits transport and deposition processes. Movement of bedload material takes place only at high discharges while equally there is generally a threshold discharge level below which no sediment movement occurs. When flow decreases to a level below this threshold, sediment is deposited and will generally remain undisturbed until a flood (of equal or greater magnitude than the threshold) recurs. The time span may be over several hours or days (downstream of hydroelectric power plants), several months (during the snowmelt period), or over several years or decades in conjunction with climatic factors. During periods of high flow, increases in sediment load comes in part from the channel itself and in part from sources beyond the main channel; consequently, channel morphology is constantly being modified through time by erosion and deposition processes (Brookes, 1995). However, erosion and deposition processes are time- and place-dependent. In the middle and lower courses of rivers, for example, deposition generally exceeds erosion. Vegetation also plays a determining role; it traps sediment and in sufficient densities can protect banks from erosion.

Research in Australia (Phillips, 1989) demonstrates that where 15% of total upland sediment was eroded 50% reached streams and became trapped and stored downstream in riparian wetlands. The diffusion/dispersion model (James, 1985) allow for a greater understanding of the transfer patterns of sediment across the floodplain when a river overflows. The major factors affecting sediment distribution are: (i) flow velocity (a decrease in velocity can result in the construction of levées at the edge of channels and complex processes throughout the floodplain, because of the existence of abandoned channels, sloughs, hollows, ridges, etc. and (ii) turbulence which affects vertical and horizontal deposition of sediment. It is known, for example, that sloughs and abandoned channels have the highest rates of sedimentation (Bhowmik and Demissie, 1989; Hupp and Morris, 1990; Sparks *et al.*, 1990; Hupp and Bazemore, 1993).

The role of vegetation on sedimentation processes

The riparian environment plays an important role in fluvial geomorphology. It influences sedimentation processes and bank stability as well as acting as a resistance to flow, especially where debris dams are formed (Hickin, 1984). Sedimentation rates have been observed to increase where riparian vegetation is present (Nanson and Beach, 1977; Hickin, 1984; Brunet and Gazelle, 1995) while deforestation is known to provoke large scale erosion. Studies involving the measurement of sedimentation in North American woody riparian wetlands (Hupp and Morris, 1990; Hupp and Bazemore, 1993) have shown a sedimentation rate varying from 0.60 cm year^{-1} in Cypress-Tupelo forests to 0.10 cm in deforested wetlands. It has also been demonstrated that invading phreatophytes such as Tamarix or Eucalyptus communities are especially effective in trapping sediments and stabilising alluvial deposits in response to changes in water regimes caused by dams (Graf, 1982; Griffin, Stafford-Smith *et al.*, 1989; Bren, 1992). By encouraging the deposition of sediment, phreatophyte communities, in turn, proliferate leading to an extension of inundation zones. These dynamic interactions, occurring on relatively short time scales, give an idea of similar processes operating in the long term.

By increasing roughness and reducing the flow velocity, vegetation favours the deposition of sediment (Pautou *et al.*, 1972; Malanson, 1993) In turn, sedimentation results in the expansion of vegetation communities. However, it has also been shown that the spatial expansion rate of pioneer plants (e.g. Phalaris, Phragmites) is checked by an increase in flow velocity when the aggrading zone reaches a critical area (Tsujimoto *et al.*, 1996). A substantial understanding of reed growth (e.g. Phragmites mauritianus) at different rates of sedimentation has allowed the development of a model which predicts sediment movement and reed growth using discrete-state variables. The model is used to predict the impact of developments in the river catchment areas or dam construction (James *et al.*, 1996). However, more precise analysis concerning the individual species, their size and physiognomy, their distribution in relation to shear stress (erosion), flow velocity and surface roughness is still needed (Malanson, 1993).

Woody debris and log-jams swept away or up-rooted during periods of high flow interact with stream morphology, storage of fine sediment and woody debris movement. Along small rivers, log-jams can build natural dams, blocking suspended material. Along larger streams, the woody debris also acts as obstacles trapping diaspores and, in so doing, initiate the development of alluvial soils and plant successions leading, eventually, to the perpetuation of fresh supplies of woody debris (Marston, 1982; Swanson *et al.*, 1982; Malanson and Butler, 1990; Montgomery *et al.*, 1996).

STRUCTURE OF RIPARIAN PLANT COMMUNITIES

Riparian plant communities are often defined by geographical (bioclimatic) boundaries. Their distribution is equally a function of the elevation of the site on which they are situated (on the intrariparial scale) and on levels of soil moisture (itself dependent on water table levels and flooding events). The spatial scale of riparian vegetation structure is further influenced on a temporal scale when events responsible for significant geomorphological change occur; sediment types, alluvial deposits, topography (i.e. the presence or absence of levées, ridges, depressions) and even the formation of the floodplain itself, constantly change through time. Riparian vegetation can in fact be seen to develop in response to such external abiotic processes over 3 temporal scales (Mitsch and Gosselink, 1993): (i) a century/a millennium with respect to changes in hydrological conditions, soil type and the stability of deposits; (ii) a decade, where wet or dry periods can affect the establishment and growth of seedlings and regenerating individuals and (iii) over a year, during which time dominating influences such as the seasonality of temperature and hydrology play an important role in shaping vegetation structure. Human impacts on riparian plant community structure generally takes place over an even shorter time scale (months, days, hours) and tend to govern biogeochemical cycles and biota (Gosselink, Brinson *et al.,* 1990; Mitsch and Gosselink, 1993).

Establishment and succession

Riparian vegetation becomes established and new successional sequences unfold wherever the river has created new sites during periods of high discharge. These new sites include (i) areas of deposition such as mid-channel islands (islets), point bars, side bars, terraces and (ii) abandoned channels and sloughs. Favourable sites for the establishment of vegetation undergo a continual process of renewal and destruction depending on flow rates. Hydrological conditions also control the release or storage of sediments and diaspores from the riparian zone (Kelsey *et al.,* 1987). These tend to move gradually downstream in pulses, becoming mobile during flooding events. Nevertheless, even as early as in the initial stages of development, some differences occur. The reasons for these differences are linked to geomorphic processes, the role of large woody debris, substrate conditions (availability of nutrients, water, organic matter) and the degree of competition between different species (Walker and Chapin, 1986; Malanson and Butler, 1990).

The successional processes are controlled by site conditions essentially dependent on flooding, sedimentation rates and autogenic processes related to competition. For example, it has been shown (Conchou and Pautou, 1987) that *Phalaris arundinacea* populations are controlled in their initial establishment and subsequent growth by sediment type, flood duration and fertility of the deposits. Generally, floods allow for rapid establishment of plants by preparing the ideal seed-beds required by pioneer species. When a site becomes isolated from active river processes and elevation provides sufficient protection from flood waters, a successional sequence begins to evolve; pioneer species *(Populus, Salix, Alnus)* are replaced by hardwoods *(Fraxinus, Acer, Quercus or Ulmus)* as the necessary abiotic conditions for each successional stage becomes available (Johnson, Burgess *et al.,* 1976; Pautou, 1980; Pautou, 1985). It is necessary to emphasise that flooding periods must coincide with periods of viable seed release. This relationship highlights the vulnerability of species such as *Salix, Populus* and *Tamarix* which are notable worldwide riparian pioneer species (Malanson and Butler, 1990). These plants become established after floods in zones which have been inundated and soon cover the site thereby increasing sedimentation in that zone by trapping of materials. This extensive storage of sediment allows for vegetative regeneration of these species since any buried twigs can give rise to new individuals. Major changes may then occur affecting successional pathways in sloughs, abandoned channels and deltas in which sedimentation, floods and an accumulation of organic matter undergo complex interactions (Waldemarson-Jensen, 1979). Within abandoned channels and sloughs characterised by fine material deposits, pioneer species are more closely linked to nutrient availability (Pautou, 1984).

COMPLEX INTERACTION OF SEDIMENTATION AND VEGETATION STRUCTURE

Several authors (Bedinger, 1979; Hupp, 1982; Hupp, 1983; Barnes, 1985; Hupp and Osterkamp, 1985; Pautou, 1989) have demonstrated that flood duration and frequency (controlled by topography) could be the major factors affecting the internal structure of riparian communities. Species associations and their niche within the floodplain could be more closely linked to these factors and to the stream energy than to physico-chemical characteristics of sediment, but it is difficult to dissociate the effects of sedimentation and flooding on vegetation structure. Without a succession of floods supplying sediment, aggradation would not occur. Furthermore, flooding, sedimentation and vegetation are influenced by other closely linked parameters:

Soil moisture

Seemingly the most important of the edaphic and climatic variables, it plays a major determining role in the survival of herbs and in the establishment phase of woody plants. It has been shown (Dawson and Ehleringer, 1991) that shrubs are dependent on surface soil moisture while larger trees rely on deeper sources of water.

Mechanical action of river flows

The force of a flood can affect the composition of the riparian vegetation through mechanical injury, deposition of fine sediment and inundation (Broadfoot and Williston, 1973). The effect depends on the age and stage of development of communities. Flood flows cause a redistribution of litter which is more or less favourable to plants (Nilsson and Grelsson, 1990; Langlade and Décamps, 1994; Lavertu *et al.,* 1994). It has been shown (Bush and Van Auken, 1984; Hupp and Osterkamp, 1985; Pautou, 1989) that the internal structure of riparian plant communities is linked to topography and flooding frequency; as a result, biodiversity increases along a gradient of elevation.

Oxygen depletion

Riparian plant species are more or less adapted to periods of oxygen depletion (Kozlowski, 1984). The duration of floods determines how anoxic conditions may become but this is also a function of the elevation, microtopography and soil heterogeneity of a site rendering a substrate more or less susceptible to drainage processes.

Organic matter content

In those systems where there is a scouring of the river bed and significant transport of materials, substrate is usually coarse. Soils are thus generally less well developed and have organic matter contents less than 2 to 5% (Mitsch and Gosselink, 1993). Alternating periods of flooding and low flow and the presence of clay slows down but does not stop the decomposition of organic matter. Nevertheless, high levels of primary productivity in the stream and large allochtonous inputs (debris) influence litter quality and quantity and, hence, vegetation structure.

Presence of nutrients

Alluvial zones are highly charged with nutrients. This is in relation to (i) the high clay content of alluvial deposits promoting the retention of phosphorus, (ii) organic matter, a rich source of nitrogen, and (iii) a regular input of nutrients during periods of high flow with significant contributions from lateral runoff (Lowrance *et al.,* 1986). Spatial and temporal retention of nutrients are closely linked to the geomorphology of catchments and channels (Marti and Sabater, 1996). Periods of anoxia triggered during successive flooding periods have reducing properties responsible for changes in pH values and hence in the mobilisation of minerals such as phosphorus, nitrogen or magnesium. Denitrification processes, however, are favoured in substrate that are successively flooded and drained (nutrients are more rapidly released in litter subjected to a succession of aerobic and anaerobic phases: Verhoeven and Van der Thoorn, 1990).

SEDIMENTATION AND ANTHROPOGENIC IMPACTS

Human activities may increase erosion, sedimentation and flooding processes. Numerous human impacts lead to siltation in channels during periods of low flow. These transient deposits of sediment occur below construction sites and as a result of agricultural practices and forest removal (Bhowmik and Demissie, 1989; Brookes, 1995; Freedman, 1995). In disturbing sedimentation rates and flow, human activities consequently affect plant succession (Pautou, Girel *et al.,* 1985; Bravard *et al.,* 1986). Succession types may be then purely abiotic (Morin *et al.,* 1989). Nevertheless, some species such as *Tamarix,* whose development is closely linked to high rates of sediment, may be favoured. The anthropogenic factors particularly influencing flow regimes are embankments. Embankments concentrate water, energy and matter in a single channel and limit floodplains to a small area between the dykes. Change in dispersion patterns of flow are noted downstream of hydroelectric power plants, the latter directly influencing the grain size of river beds and island deposits (Dietrich *et al.*, 1989).

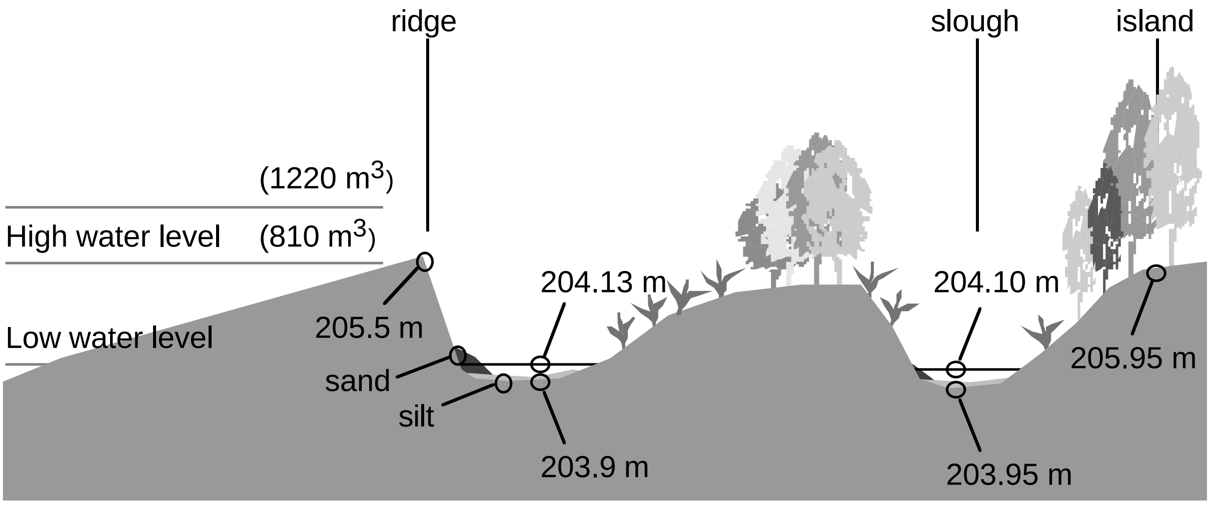
In a recent study on the Colorado River, USA (Steven *et al.*, 1995) a list of major disturbances caused by reservoirs was assembled. Downstream from dams, daily fluctuations relating to hydroelectric power generation resulted in a tidal zone devoid of vegetation and an overall decrease in sediment transport in these areas. However, eddies which occurred along the length of the river encouraged sedimentation in these areas where flow is much reduced. While these deposits are not very stable, they tend to be very persistent and are colonised by tall-grasses (e.g. *Typha, Phragmites, Juncus*) on silty deposits and by woody brushes (Salt-Cedar and Willows) on sandy bars. Factors controlling vegetation were flooding frequency, soil texture and distance from the dam.

SEDIMENTATION AND VEGETATION STRUCTURE IN THE LARGE ALPINE FLOODPLAINS

The first case study is taken from the French Upper Rhône river valley. The site, Brégnier-Cordon at the southern tip of the Jura Mountains (Fig. 2), is an ancient old lake basin filled with sand and clay. It covers an area with a shallow gradient (0.50 to 0.55 m km^{-1}, maximum width: 6 km) through which a meandering channel pattern has evolved. During the 19th century, the construction of an overtopping dyke to facilitate navigation has lead to the development of a braided/anastomosed floodplain system in which sedimentation progresses in 3 main stages: (i) formation of natural levées on banks (height 2 to 3 m), stabilisation of the landform by shrubby willow *(Salix purpurea, S. triandra, S.alba)* stands and subsequent establishment of alder *(Alnus incana)* woodlands; (ii) deposition of fine materials (silt and clay) on the islands and colonisation by oak/ash/poplar *(Quercus robur, Fraxinus excelsior, Populus alba)* mixed forest characterising the alluvial shelves and (iii) formation of sloughs which are progressively filled by fine material and then colonised by an elm/willow *(Ulmus minor, Salix cinerea)* community (Fig. 3).

.

Figure 3. Transverse structure of alluvial deposits along the French Upper Rhône (91.5 km).



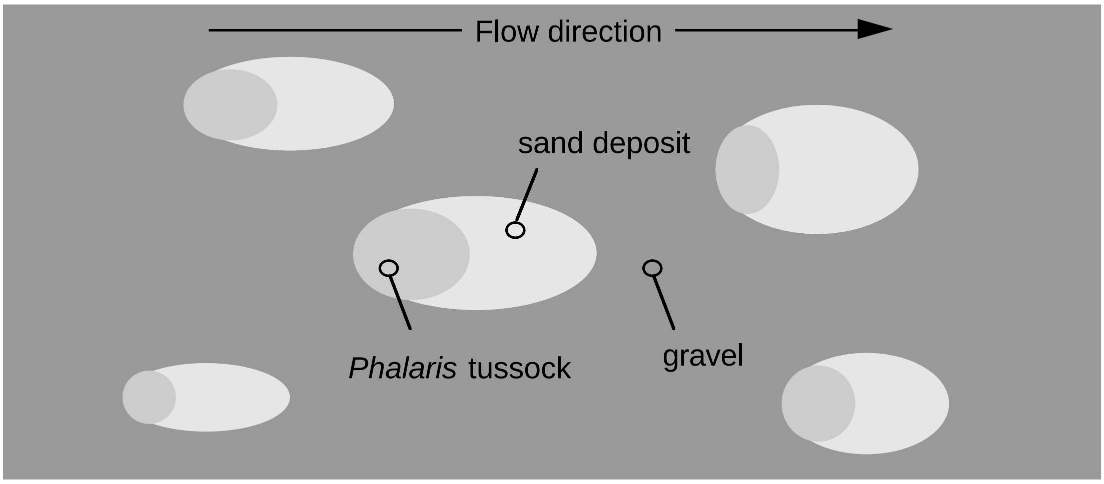
During periods of high flow, the ecological parameters of sandy deposits and anastomosed channels are modified. Factors altering vegetation structure include (Pautou *et al.*, 1972; Pautou, 1979): (i) vertical spread of the oldest deposits, (ii) filling of sloughs and hollows and subsequent coalescence of islands which were separated by secondary abandoned channels beforehand, (iii) establishment of fine grained platforms (rich in gleyed colloids) at the downstream ends of islands, (iv) silting linked to a lateral shift of main and secondary channels, (v) sedimentation of bedload materials on the upstream tip of islands (vi) filling of areas of confluence due to a decrease in flow velocity and in the formation of eddies, (vii) emergence of new alluvial bars and (viii) deposition of natural levées along secondary channels.

The early stages of the sequence

The coarse materials of bedload (pebbles, gravel, coarse sand) are deposited first and constitute dunes when the flow discharge exceeds 900 $m^3 s^{-1}$. The dune roof is overtopped when the discharge exceeds 810 $m^3 s^{-1}$ (mean discharge: 400 $m^3 s^{-1}$). The fine suspended load is deposited on the gravely substratum when the discharge is less than 500 $m^3 s^{-1}$.

On the middle part of the bar, a community of *Phalaris arundinacea* can establish. This reed is anchored in the substratum by its root system. By increasing the bed rugosity, it initiates sedimentation of sand which deposit behind each reed clump (Fig. 4). The next year, these dunes increase in length and volume. The processes generally stop when the dunes reach a critical volume (Tsujimoto *et al.*, 1996). The biggest dunes are 2 m long and 0.60 m deep. New clumps supporting perennial grasses such as *Calamagrostis littorea, Agrostis alba* and *Festuca arundinacea* generally come in between. This colonisation induces a typical microtopography characterised by the presence of ridges and hollows. Within the hollows, silt and organic matter are trapped; nitrophilous plants such as *Polygonum lapathifolium, Ranunculus repens, Deschampsia caespitosa, Mentha aquatica* or *Bidens tripartitus* grow on these eutrophic sites.

Figure 4. Sediment trapping by *Phalaris arundinacea.*



On the less elevated parts of the bars, the deposits are colonised by a *Phalaris arundinacea* community which promotes a rising of the ground level by sediment trapping. The shade effect produced by heliophilous and hygrophilous plants will allow the subsequent establishment of sciophilous and mesohygrophilous species.

In both cases, the arrival of willows *(Salix alba, S. triandra, S. viminalis, S. purpurea, S. daphnoides)* and poplars *(Populus alba, P. nigra)* is recorded. The softwood shrubby thickets become rapidly dense and the herbaceous layer disappears.

The lateral channel shift explains the presence of water bodies between the bars. These hollows or swales (the so-called "lône") are at first open water habitats without vegetation; then, once the filling by fine sediments has begun and flooding is no longer permanent, *Typha latifolia, Phragmites australis, Carex acutiformis, Veronica beccabunga, V. anagallis* become established. Some islands do not evolve beyond this stage which can be reached after 15/20 years.

If sedimentation can progress, the willow community (*Salix alba* woodland) colonises the whole island over several years. This community, which characterises soils little affected by superficial waterlogging (presence of *Cornus sanguinea, Ligustrum vulgare, Impatiens glandulifera, Brachypodium sylvaticum* in the understoreys), will evolve to a mature phase, then to a decline phase in which, alders *(Alnus incana)* will replace willows in the gaps.

The middle stages of the sequences

Heterogeneity of habitats is increased by deposition of sediment and coalescence of islands. Three habitat types occur: (i) the sandy levées along the banks, (ii) the central island basins where flood water can stagnate for long periods (the zone can be subdivided into terrestrial sandy platforms, at the upstream tip and a silty/clayed semi-aquatic/aquatic area at the downstream tip) and (iii) the swales (or sloughs) filled by clay and silt and isolated from the main channel.

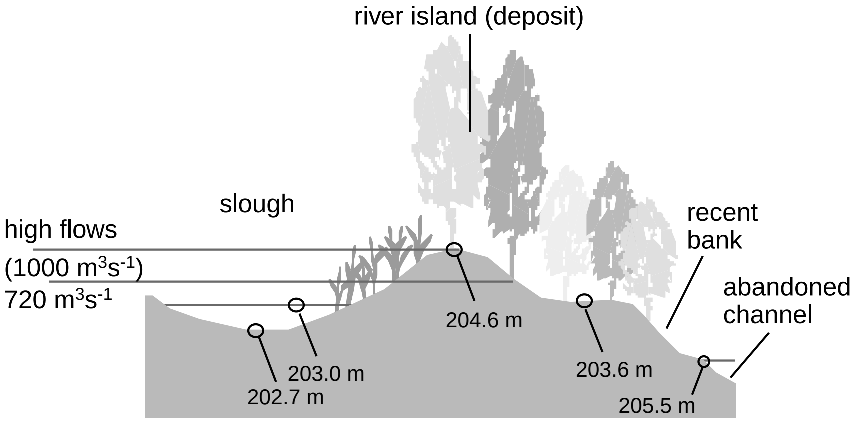
At this evolutionary stage, islands support a grey alder woodland *(Alnus incana – Equisetum hiemale* community). Some ashes *(Fraxinus excelsior)* may appear on the more elevated levées.

Grey alders thrive at the upstream parts where a thin silt layer lies on deep sandy deposits. The alluvial soil is permeable and supports species avoiding superficial waterlogging such as *Crataegus monogyna, Ligustrum vulgare* or *Evonymus europaeus.* Some shrubs characteristics of hardwood forests *(Fraxinus excelsior, Quercus robur)* may appear and the herbaceous flora is represented by *Rubus caesius, Impatiens glandulifera, Humulus lupulus, Tamus communis, Galeopsis tetrahit* and *Circaea lutetiana.* In the downstream areas, the fine deposits, flooded during long periods, support a mixed softwood community *(Alnus incana, Alnus glutinosa, Salix alba).* The silty/clayed swales exhibit, from downstream to upstream, a reed community *(Phragmites australis),* a reed/sedge community *(Phragmites australis, Phalaris arundinacea, Carex riparia, Carex acutiformis, Carex pendula),* a sedge community *(Carex acutiformis)* and, on deep silty deposits (0.30 to 0.50 m), a wet meadow *(Ranunculus repens)* invaded by willows. When the flow discharge reaches 450/500 $m^3 s^{-1}$, the latter community is flooded; this explains the presence of hygrophilous and mesohygrophilous plants well adapted to fine deposits such as *Lysimachia nummularia, Myosotis palustris* and *Ranunculus repens.* When the clay-plug reaches 0.80 m high, it prevents water from flowing to the river and favours the establishment of a stagnant water body which is filled slowly by fine sediments and sludge. On deep silty layers which sometimes exceed 1.50 m, only elms *(Ulmus minor)* and willows *(Salix cinerea)* grow; they constitute a dense shrubby community without a herbaceous layer.

Evolution to alluvial hardwood forests

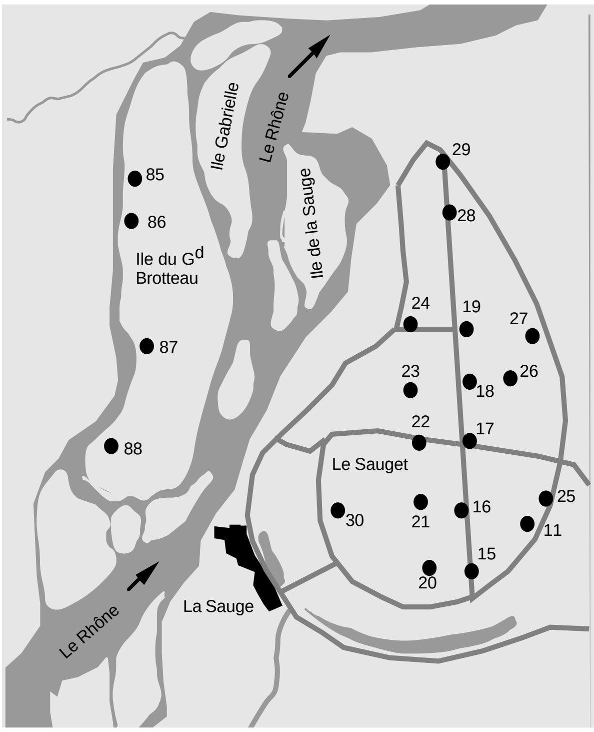
As soon as sedimentation processes develop, the alder woodlands colonise from upstream to downstream. At the same time, oaks *(Quercus robur)* emerge as indicators of evolution towards a hardwood forest. Sedimentation caused a vertical accretion and a lateral growth in the riparian deposits. The formation of recent banks made of silty material is observed; they are colonised by *Phragmites australis, Typha latifolia* and *Phalaris arundinacea* which, in turn, will be rapidly replaced by willow bushes *(Salix triandra)* (Fig. 5).

Figure 5. Cross section of a river island (French Upper Rhône river, 87 km, île gabrielle).

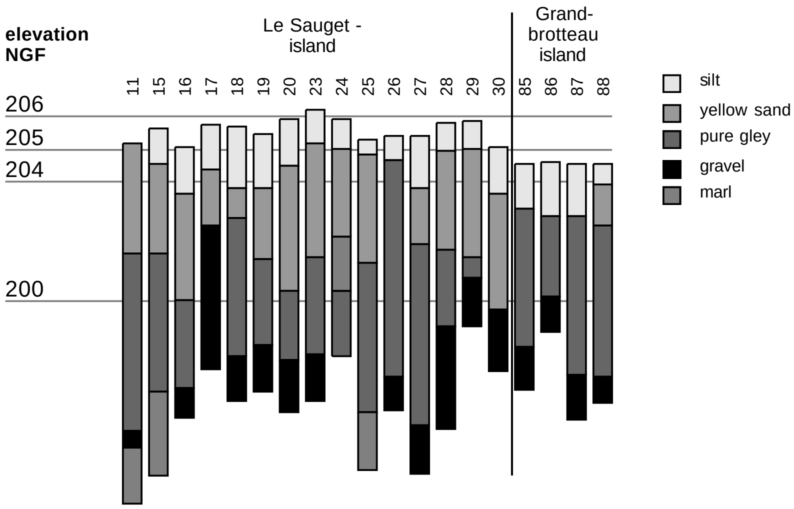


At this stage, any raising of the island level can be only made during exceptional flood periods. In this case, sedimentation processes will deposit mainly silty materials. On an old island such as Le Sauget in Bregnier-Cordon site (Fig. 6a), which already existed in 1607 (De Beins-De Dainville mapping), the silt layers always reach 1.40 m (maximum 2.40 m) whereas the sandy layers lying beneath reach 3.5 to 5 m high at the bank levées and 1.00 to 2.00 in the central part (Fig. 6b). The island elevation corresponds to the level of a 50-year flood (1450m^3 s^{-1}). When they are not farmed, these islands are characterised by hardwood species such as *Quercus robur, Fraxinus excelsior, Juglans regia, Acer pseudoplatanus* and *Ulmus minor.* Shrubs such as *Ligustrum vulgare, Lonicera xylosteum, Crataegus monogyna, Evonymus europeus* and herbaceous plants such as *Hedera helix, Circaea lutetiana, Arum maculatum, Paris quadrifolia* or *Galeopsis tetrahit* are present in the understorey. The bottoms are characterised by *Ulmus minor* and *Lysimachia nummularia* whereas the most drained deposits support *Quercus robur, Paris quadrifolia* and *Polygonatum officinale.*

Figure 6. Sedimentation levels on two islands in French Upper Rhône river.



a) location of sequences

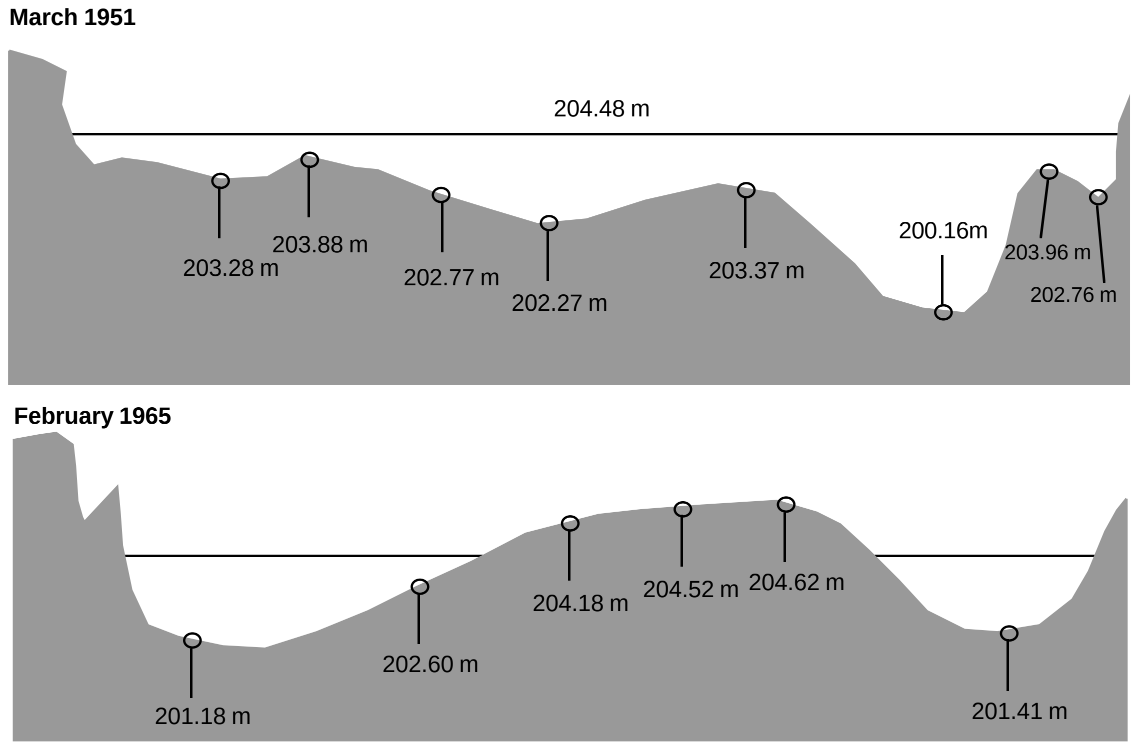


b) sediment sequences

In swales and in filled abandoned channels where there is a grain size gradient (gravel to silt) from upstream to downstream, the last sedimentation phases lead to alder *(Alnus glutinosa)* woodland then to the alder/ash forest.

It is important to underline that some islands can form if the flow velocity decreases in the main channel (Fig. 7); this type of mid-channel island changes laterally and from upstream to downstream but, in this case, no swales are produced.

Figure 7. Development of a mid-channel island (French Upper Rhône river, 90.5 km).



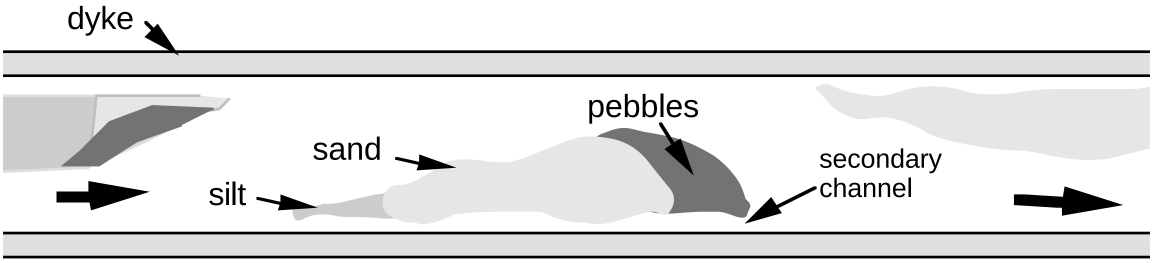
SEDIMENTATION AND VEGETATION STRUCTURE ALONG CHANNELISED RIVERS

For the most part, Alpine streams are straight and channelised, constrained by dykes and hence have a considerably reduced floodplain (Girel, 1993; Girel, 1994a; Girel, 1994b). These disturbances have greatly affected flow dispersion patterns, creating a new set of ecological conditions with regards to sedimentation, flooding and vegetation coverage. Other more recent disturbances of anthropogenic origins such as dredging, gravel mining, damming and pumping of water for irrigation have further impacted natural patterns of flow and sedimentation. The relationship between vegetation and sedimentation will be studied both within the dyked channel and outside it, in the abandoned floodplain where farmed soils have been made by artificial silting during the 19th century (Girel, 1994a; Girel, 1994b).

Sedimentation and vegetation within dyked channels

A dyked channel is similar to a ‘circuit’ in which large volumes of sediments, woody debris, nutrients and dissolved organic matters travel and where there is a high energy during exceptional flooding events. Dyked streams function like any other straight/constrained systems in which the slope value allows for the deposition of sediment (Petts and Foster, 1985; Brookes, 1988). Sediment moves both from upstream to downstream reaches and from one bank to the other. Lateral bars are formed and are separated from the dyked bank by narrow secondary channels, flooded only during periods of high flow (Fig. 8). These side-bars generally alternate from one side of the river to the other and are separated from each other by a distance which is equal to 4-6 times the channel width (Knighton, 1984). During periods of high discharge, lateral bars may move by gradual erosion of the upstream tip and construction (deposition) on the downstream end. In a dyked system which is not disturbed by new perturbations the "life cycle" of the deposit and consequently of the plant succession which develops are limited over time. However, sedimentation is a process varying according to: (i) the type of stream, the presence of different reaches in the stream beds, the volume of water flowing during the annual cycle and (ii) engineering works which have developed along the length of the river.

Figure 8. Shifting islands along an artificially straightened river (the dyked Isère River between Albertville and Grenoble).



Examples of the Isère, Drac and Romanche rivers (Upper French Rhône system, see Fig. 2) have been chosen, all of which have been dyked during the 19th century. The first consequence of the dykes was an accumulation of huge volumes of sediment in certain reaches and subsequent flooding due to breaches of the embankments. Dredging and gravel extraction activities counteracted aggradation processes along with the progressive incision of the river bed provoked directly by the straightening of channels. Also associated with channelisation are drops in water table levels (2-3 metres). These processes have been followed by changes to vegetation communities. Post-pioneer communities such as alder *(Alnus incana)* and hardwood woodlands *(Fraxinus excelsior, Corylus avellana)* become established on bars in which the last stages of succession do not normally pass the willow brush stage. Further development leads to the establishment of a hardwood community *(Quercus robur, Fraxinus excelsior, Ulmus minor, Robinia pseudacacia).* The construction of hydroelectric reservoirs has resulted in the trapping of sediment and has promoted incision below the dams. Damming also affects energy- and water-fluxes. On the other hand, along the Isère river, high flow discharges persist from November to June, during wet periods (e.g. "rain on snow" periods in November or February). As a result, sedimentation becomes very important (i.e. during the year 1994/95, sandy deposits exceeded 0.50 m on lateral bars upstream of Grenoble; Pautou *et al.*, 1996).

Close relationships exist between community types, floristic composition and levels of sediment deposits. In the pebble layer located at the upstream tip of bars (lowest elevation) plants can only become established when sheltered by obstacles such as boulders or debris dams. In the case of the Isère, Drac and Romanche rivers, the pioneer which plays the major role on the geomorphological processes is the grass *Agrostis stolonifera.* The adventitious roots soon spin a web anchoring the grass to gravel deposit while above-ground vegetation traps sediments. Reeds *(Phalaris arundinacea, Calamagrostis littorea)* play a similar role when elongated dunes are gradually formed. Flow velocity is also slowed down by the deposits, thus increasing sedimentation of silt on the tops of the dunes which are then colonised by reeds and grasses followed by shrubby willows such as *Salix alba, S. triandra, S. purpurea, S. viminalis* and *S. daphnoides.*

Presently, the most developed communities, which are less than 10 years old, are willow bushes colonising silt/sand bars of elevations of 1.00 to 1.50 metres. Grey alder *(Alnus incana)* white poplar *(Populus alba),* black locust *(Robinia pseudacacia)* and maple *(Acer pseudoplatanus)* begin to establish in place of these communities (particularly on the banks of secondary channels) and one might even ask if the alder community stage will not be by-passed. The absence of ecological constraints linked to important flooding events could directly favour the establishment of hardwood species.

The plant succession stages can be summarised as follows: bare substrates (pebbles) to *Agrostis stolonifera* to *Phalaris arundinacea* to *Phalaris arundinacea* and *Phragmites australis* to willow bushes to *Salix alba* woodland to ash/maple woodland. The last stage depends on whether flow discharges are great enough to allow for the continuation of sedimentation processes.

Along the Drac river, side-bars are different due to the absence of high discharges. Colonisation patterns of the upper reaches of the side bars remain constant, with *Agrostis stolonifera* playing the major role. On more elevated deposits, willows *(Salix purpurea, S. eleagnos)* and black poplar *(Populus nigra)* are the first to become established. However, their growth is only possible if the flow level remains high enough for a sufficiently long period to allow for the establishment of the seedlings. If the water level drops too quickly, the seedlings will not persist. By trapping sediment, willow saplings play the same role as reeds. The depth of the sediment layers increases from upstream to downstream reaches (it can reach depths of 1.00 m in the middle part of the side-bar and 1.50 m at the lowest top). The elevation increase induces modifications concerning the biotic factors: (i) increase in herbaceous and woody plants biodiversity, (ii) increase in woody plants height, (iii) increase in number of age-classes concerning trees (the oldest poplars are more than 35 years old), (iv) increase in plant covers and particularly in the herbaceous layer which makes an unbroken carpet in the middle part of the island and (v) increase in number of plants characterising lowland and upland hardwood forests.

Sedimentation processes are promoted by the density of pioneer plants, by the formation of log-jams and by the presence of coarse materials stored in channels before the development of hydroelectric dams. The communities which succeed one another on the floodplain are:

Agrostis stolonifera to *Populus nigra* and *Salix eleagnos* and *S. Purpurea* stands to *Populus nigra* thickets to *Populus nigra, Populus alba, Robinia pseudacaia* woodlands to *Fraxinus excelsior, Tilia cordata, Acer pseudoplatanus, A. platanoides, Carpinus betulus.*

Sedimentation and vegetation outside the dyked systems

The construction of dykes has prevented large areas of lowland from being flooded during periods of high flow. These areas, originally of braided/anastomosed form, are typically constituted of coarse alluvial deposits (sand and gravel) and were made up of numerous channels. In order to fill these lowlands and to create agricultural land, hydraulic engineers used an old method already used in previous centuries by the Incas in the Andean floodplains (Zimmerer, 1995) and also by Italians since the 17th century (Alexander, 1984). This old method ('warping' or 'colmata') involved making use of suspended load of high flows (from April to August) in order to fill and reclaim land for farming on the floodplains (Girel, 1994b). The area to be filled was subdivided into decantation basins (Fig. 9) and in this system of lagoons, water flowed slowly from one basin to another, depositing suspended sediment along its course. The largest particles (fine sand) are deposited first while the finest (silt and clay) are only to be found at the bottom of the system (Fig. 10).

Sardinian and French engineers managing the floodplains along tributaries of the Rhône river describe in numerous reports published between 1830 and 1870 interesting observations with regards to the stages of colonisation in warped basins (Girel, 1994a). These new habitats were first colonised by plants having efficient dispersal abilities either thanks to air- or water-flow. The first warping, performed on gravel deposits and secondary channels, supplied a thin sediment layer (0.10 to 0.02 m from upstream to downstream). These silty habitats were colonised during the following year by aquatic plants and mainly by the *Typha. Typha latifolia* and *Typha angustifolia* were probably present; however, *Typha minima* (an Alpine alluvial plant still present nowadays along the river Isère between Albertville and Grenoble), probably constituted the major population.

Figure 9. Artificial filling of old braided channels and gravel deposits: the warping systems used along the Isère River downstream of Albertville.

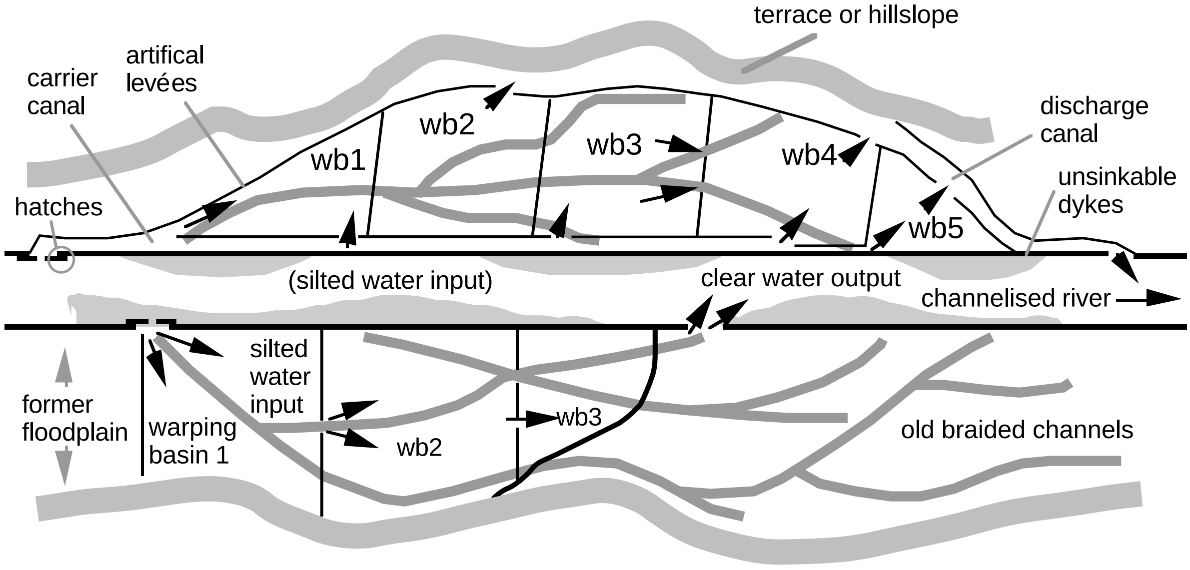
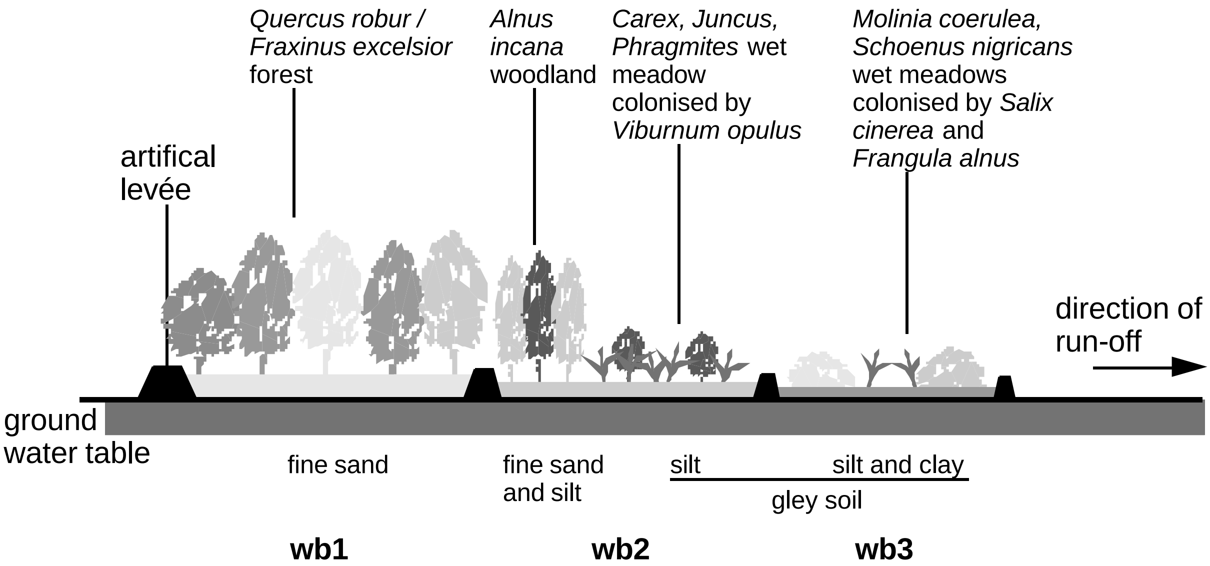


Figure 10. Soils and vegetation in a warped floodplain (the Isère River downstream of Albertville).



After three years of warping, reeds *(Phragmites communis)* spread throughout the warped basins. During the 4th and 5th years, while the elevation of sites are raised by sediment trapping and the formation of a thick mat of rhizome, the health of the reeds vegetation improves. The rate of hard stems without leaves characterising the semi-aquatic and aquatic vegetation types decrease and a biomass suitable for cattle litter results. During the following years, a dense community of sedges *(Carex maxima, C. paludosa, C. riparia, C. stricta)* can develop. At this stage, the herbaceous layer traps a great part of the suspended load, releasing very clear-flowing water in discharge canals.

If the warping was stopped at this stage, a wet meadow was obtained. Generally, its soil was just above the level of the water-table (at 0.05 to 0.20 m depth during periods of high flow). Successive mowing (two harvests between June and August), grazing (during autumn and winter) without replacing lost nutrients (the basins are no longer supplied in matter and nutrients due to the presence of embankments) and cattle trampling caused vegetation changes. Reeds are replaced by sedges, grasses and hygrophilous dicotyledons. Thus, the fens which produced mainly litter were replaced, after several years, by mowed and grazed wet-grasslands in which *Schoenus nigricans, Molinia coerulea, Juncus* and *Carex* were dominant, depending on the depth of the silt layer.

Definitions of plant succession in warped basins contributed by 19th century engineers are strengthened by a recent study carried out in the Reuss delta, Switzerland (Leuthold, 1994). Following the occurrence of an exceptional flooding event, the effects of siltation on wet meadows were recorded. The study highlighted the fact that the deposition of silt promotes the proliferation of root-systems which are superficial and very spread out, such as occurs with *Phragmites australis, Equisetum palustre, Juncus articulatus* or *Agrostis gigantea.* The disturbance caused the elimination of several species (such as orchids); however, some did re-establish later. Some new species may also appear; nevertheless their establishment was not permanent. The persistence of the plant communities depends on the depth of the silt layer deposited. Plants such as *Phragmites australis, Molinia caerulea* and *Carex elata* are less affected and hence are present in most of the communities. Generally, the elimination of the *Schoenus nigricans* community was noted in the site where the silt depth exceeded 0.05 m. It was replaced by a *Carex, Stachys, Molinia* fen from 0.05 to 0.15 m and by a *Valeriana dioïca-Filipendula ulmaria* wet-meadow when the silt layer exceeded 0.20/0.30 m depth.

In the case of *Molinia coerulea-Schoenus nigricans* community, it is known that the successive mowings at the end of summer inhibit the formation of flowered fertile stems of *Molinia* and, hence, limits its development; *Schoenus nigricans* and the other species can then spread over (Schopp-Guth *et al.,* 1994). On the other hand, drainage is favourable to *Molinia* which is also well adapted to soils poor in organic matter and affected by high variation ranges of the ground water table (Meade and Blackstock, 1988). *Molinia coerulea* could be affected by recurrent flooding and its speedy establishment after drainage could be linked mainly to the elimination of the superficial waterlogging; the presence of *Schoenus nigricans* could be closely related to a high Ca^{++} level (Gilman, 1994).

Abandonment of mowing practices, development of drainage and impacts of high variations in the water-table linked to the Isère river could explain the wide extension of *Molinia coerulea* communities on the warped basins.

Since the end of the Second World War, the wetlands formed by warping have become derelict. As a result, succession could take place and there have been several models depending on the type of warping and on the duration of warping operations (Girel, 1994a). Thus, in the "successive discharge system" (see Fig. 8, right bank of the river) and in "simultaneous feeding system" (Fig. 8, on the left bank), the duration of warping operations controlled the depth of material deposited and, partly, the depth of the water table. In the first case it will be noted that the depth of the sedimentation and the grain size of the particles decrease along a gradient from upstream to downstream. These parameters explain the presence of well-developed communities (ash/oak/lime hardwood forests) on deep sandy deposits, the presence of Alder woodlands on silty/sandy deposits or the presence of *Molinia coerulea/Schoenus nigricans* on superficial silty/clayed soils (Fig. 10).

DISCUSSION AND CONCLUSION

Large intra- and perialpine floodplains are the result of huge sedimentation processes which led to the filling of glacial depressions. This deposition was not continuous. In the case of the Rhône river, the old post-glacial "Lac du Bourget" (see Fig. 2) has conserved its aquatic characteristics. Hence, one can explain the juxtaposition of a large range of communities linked to both mineral alluvial soils and organic soils originating from a long process of terrestrialisation. Neolithic, Post-Roman and Medieval deforestation of watersheds have increased the volume of material- and water-runoffs and, as a result, have induced changes leading to a floodplain with various elevations. The most elevated sites are suitable for hardwood forests and farmlands whereas the less elevated support wetlands and softwood forests.

Civil engineering works which began at the end of the 17th century triggered a homogeneisation of floodplain forms but conversely differences concerning sedimentation patterns. Embankment procedures, in particular, are at the origin of: (i) a concentration of sediment in channels, (ii) a rapid raising of islets and (iii) a rapid siltation in sloughs and secondary channels. It has been shown (Moore and Evans, 1991; Pautou *et al.,* 1996) that sedimentation of alluvial plugs, the establishment of helophyte communities in abandoned channels and the aggradation of silty/sandy layers on the gravel substratum of islands all contribute to the formation of wetland communities (sedge, stands of willow, stands of alder) in areas where water becomes stagnant.

Thus, sedimentation processes are at the origin of altitudinal, hydrological and pedological heterogeneities. As a result, two major environmental consequences can be noted: (i) an increase in plant biodiversity and (ii) an increase in mosquito populations (Pautou *et al.*, 1995). Islands which are created by sediment deposition and lateral migration of the channels along anastomosed and braided reaches and side bars developing along dyked rivers can be considered as "buffer zones". Nevertheless, unlike riparian fringes which generally filter land run-off before it enters the stream (or other water body), river islands control fluxes of sediment, nutrients, chemical particles, pathogens, symbiotic fungi; all of which have already been introduced into the channel. Hence, they have an interesting additional role which justifies an interest in their restoration and conservation. Nevertheless, their presence is also closely linked to hydrological and geomorphological dynamics characteristic to the river. Anthropogenic perturbations which affect the stream corridor and the drainage basin can alter the functioning of the hydrosystem and consequently change erosion and deposition rates at the origin of sediment deposition processes. In this way, riparian vegetation communities may be altered and finally threatened with extinction. Additional changes are the result of the development of hydroelectric power plants where on the whole significant decreases of flow inputs are observed in the by-passed channels. Phreatophytes can then immediately invade these river stretches accelerating the sedimentation process and density of vegetation coverage thereby creating environmental problems. The example of the Drac and Var rivers reveal new types of disturbances to natural processes of sedimentation. Along the lower reaches of these streams, the cumulative impacts of civil-engineering works are responsible for changes to patterns of vegetation establishment in the channel. The perpetuation of *Populus, Ulmus, Fraxinus* and *Ostrya* thickets are now commonly observed in these reaches. The subsequent aggradation of alluvial deposits and, during periods of high flow, the uprooting of a significant number of trees creating log-jams, puts a very real strain on weirs, dykes and bridges rendering them susceptible to serious damage. While it is evident that the development of woody vegetation is in response to a much-reduced flow rate within the channel, the influence of hydrological, geomorphological and hydrochemical factors on the new patterns of vegetation establishment are still unknown. Studies that are presently being undertaken along the Isère river (ECC Programme "Abiotic controls of the establishment of woody riparian vegetation") suggest that all river reaches do not hold the same potential for vegetation.

One question remains without an answer: what might the consequences be in the case of a very significant flood (e.g. a 100-year flooding event)? In order to allow for the passage of flows and to avoid catastrophic inundations during periods of flooding, vegetation in the by-passed channels are frequently cut-back and the section is ploughed. A true successional development of vegetation cannot therefore develop on the channels and islands; softwoods are replaced by bare sediment or sparsely distributed communities of pioneers plants which cannot act as efficient buffer zones. It would seem advisable to manage riparian landscapes by careful planning (i.e. tree-removal in well-defined cycles). In the majority of cases, it will never be possible to re-establish rivers to their "natural" state (Peterken and Hughes, 1995); if efforts are to be made, any realistic restoration will only be partial.

Along dyked rivers, a lowering of the river bed (sometimes by up to several metres) has been observed over the past 10 years. This phenomenon is caused by entrenchment processes affecting reaches in which the material input (deposition) is in deficit with respect to the output (erosion). This progressive lowering of the river bed level can be explained in conjunction with several factors:

(i) milder climatic conditions on a global scale since the end of the "Little Ice Age", (ii) the recolonisation of farmed hillslopes by vegetation since the end of World War I, (iii) the development of hydroelectric dams and reservoirs from the end of World War II which changed discharges regime and trapped sediments and (iv) instream gravel extraction which has triggered a regression of erosion since the 1960s. The incision process is responsible for a significant lowering of water table levels, both in the channels and beyond the dykes, leading to a change in vegetation community structure. On side bars, the natural processes of erosion/aggradation only occur during strong discharge periods so that the successional development of riparian vegetation is able to progress beyond the early softwood stage (alder/willow/poplar thickets), usually along the most elevated deposits. Thus, ageing of the alder *(Alnus incana)* and willow *(Salix alba)* populations and the establishment of the locust *(Robinia pseudacacia)* have been observed. The latter is characterised by rapid growth on alluvial soils but is frequently up-rooted during the exceptional floods. Thus, clearcutting of woodland is carried out regularly to avoid the generation of large log-jams which might become serious obstacles to flow during flooding events.

Outside of the dykes, a lowering of water table levels leads to changes in the structure of riparian vegetation communities. The early successional woody species alder/willow thickets are replaced by ash/oak *(Fraxinus excelsior, Quercus robur)* woodlands with a gradual colonisation by several non-alluvial trees such as *Carpinus betulus, Robinia pseudacacia, Quercus pubescens, Buxus sempervirens, Acer campestre, Acer pseudoplatanus* or *Laburnum anagyroides* coming from the adjacent highlands. Ancient warping basins which could not be farmed in the past due to waterlogging are also affected by changes of vegetation as water tables are lowered; if the water table continues to drop, these zones may be drained with the loss of woodlands to farming and urbanisation.

The preceding discussion has shown that restoration and conservation of riparian plant communities as buffer zones are closely dependent on the management practices of both the river and its catchment in its entirety. First, it seems necessary to stop incision mainly by sensitively controlling flow of water and material (timing and volume of water released by dams, timing of the emptying of reservoirs) and also by the controlling water table variations (construction of transverse weirs across the main channel).

The ancient system of warping may be considered as an interesting solution as it has already supplied much information about artificial silting in relation to plant successions. This technique might in fact be used again along small tributaries flowing through highly farmed or urbanised areas. Their construction would allow for the storage of a significant volume of fine particles thereby recreating those habitats suited to certain alluvial plant communities and additionally by acting as buffer zones. Nevertheless, it must be emphasised that the restoration of such zones in conjunction with the aforementioned factors resulting in retention of sediment may aggravate the problems of incision and have an impact on water table levels and riparian landscapes. This eventuality is considered in order to underline the fact that the restoration of buffer fringes along floodplains are linked to external perturbations whose impacts must be studied throughout the drainage basin (Latulippe and Peiry, 1996). Catastrophic floods occurring after summer storms over the past few years in France, Spain and Italy highlight the necessity of restoring large buffer zones in uplands. Precipitation must be retained upstream within fens and wet meadows which constitute, like river islands and side bars, a kind of safety valve. Thorough studies are necessary in order to develop buffer zones maintaining the processes of retention, purification, conservation and production.

Acknowledgements
We thank Nadia Barsoum (Department of Geography, University of Cambridge, UK) who corrected the English version of this paper. We also wish to thank Jean-Pierre Guichard (Centre de Biologie Alpine, Grenoble, France) for his help in reproduction of the illustrations.

This work was supported by the ECC project EV5V-CT93-0261 on the 'Abiotic controls of the germination, establishment and species distribution of woody riparian vegetation'.

REFERENCES

Alexander, D. (1984) The reclamation of Val-di-Chiana (Tuscany). *Annals of the Association of American Geographers,* **74**(4), 527-550.

Barnes, W.J. (1985) Population dynamics of woody plants on a river island. *Canadian Journal of Botany,* **63**, 647-655.

Bedinger, M.S. (1979) Hydrology of bottomland hardwood forests of the Mississippi embayment. In: Clark, J.R. and Benforado, J. (Eds.) *Wetlands of bottomland hardwood forests.* New York, Elsevier. pp 161-178.

Bhowmik, N.G. and Demissie, M. (1989) Sedimentation in the Illinois River valley and backwater lakes. *Journal of Hydrology,* **105**, 187-195.

Bravard, J.P., Amoros, C. and Pautou, G. (1986) Impact of civil engineering works on the successions of communities in a fluvial system. *Oikos (Copenhagen),* **47**, 92-111.

Bren, L.J. (1992) Tree invasion of an intermittent wetland in relation to changes in the flooding frequency of the river Murray, Australia. *Australian Journal of Ecology,* **17**(4), 395-408.

Broadfoot, W.M. and Williston, H.L. (1973) Flooding effects on Southern forests. *Journal of Forestry,* **71**, 584-587.

Brookes, A. (1988) *Channelized Rivers.* Perspectives for Environmental Management. Chichester, John Wiley & Sons.

Brookes, A. (1995) The Importance of High Flows for Riverine Environments. In: Harper, D.M. and Ferguson, A.J.D. (Eds.) *The Ecological Basis for River Management.* Chichester, John Wiley. pp 33-49.

Brunet, R.C. and Gazelle, F. (1995) Alternance des phénomènes d'érosion et de rétention de la matière dans la zone inondable de l'Adour au cours d'une saison hydrologique. *Acta Oecologica,* **16**(3), 331-349.

Bush, J.K. and Van Auken, O.W. (1984) Woody-species composition of the upper San Antonio River gallery forest. *Texas Journal of Science,* **36**, 139-148.

Conchou, O. and Pautou, G. (1987) Modes of colonization of an heterogenous alluvial area on the edge of the Garonne river by Phalaris arundinacea L. *Regulated Rivers,* **1**, 37-48.

Dawson, T.E. and Ehleringer, J.M. (1991) Streamside trees that do not use stream water. *Nature,* **350** (28 March 1991), 335-337.

Dietrich, W.E., Kirchner, J.W., Ikeda, H. and Iseya, F. (1989) Sediment supply and the development of the coarse surface layer in gravel-bedded rivers. *Nature,* **340** (20 July 1989), 215-217.

Forman, R.T.T. (1995) *Land mosaics; The ecology of landscapes and regions.* Cambridge, Cambridge University Press.

Freedman, B. (1995) *Environmental Ecology (The Ecological Effects of Pollution, Disturbance and other Stresses).* San Diego, Academic Press.

Gilman, K. (1994) Cors Erddreiniog Anglesey a case study of wetland conservation (North Wales). In: Patten, B.C. (Ed.) *Wetlands and Shallow Continental Waterbodies.* The Hague, SPB Academic publish. pp 439-456.

Girel, J. (1993) Endiguements et colmatages sardes dans la vallée de l'Isère à l'aval d'Albertville. In: Piquet, F. (Ed.) *Le Fleuve et ses Métamorphoses.* Paris, Didier Erudition. pp 217-232.

Girel, J. (1994a) Old distribution Procedures of Both Water and Matter Fluxes in Floodplains of Western Europe: Impact on Present Vegetation. *Environmental Management (New York),* **18**(2), 203-221.

Girel, J. (1994b) Les aménagements du 19ème siècle dans les basses vallées de la Durance et du Var (France); impacts sur l'écologie du paysage. *Etudes Vauclusiennes,* **5**, 37-42.

Gosselink, J.G., Brinson, M.M., Lee, L.C. and Auble, G.T. (1990) Human activities and ecological processes in bottomland hardwood systems, the report of the ecosystem group. In: Gosselink, J.G., Lee, L.C. and Muir, T.A. (Eds.) *Ecological Processes and Cumulative Impacts illustated by Bottomland Hardwood Wetland Ecosystems*. Chelsea, Lewis Publishers. pp 549-600.

Graf, W.L. (1982) Tamarisk and river-channel management. *Environmental Management,* **6**, 283-296.

Griffin, G.F., Stafford-Smith, D.M., Morton, S.R., Allan, G.E., Masters, K.A. and Preece, N. (1989) Status and implications of the invasion of Tamarisk (Tamarix aphylla) on the Finke river, Northern Territories, Australia. *Journal of Environmental Management,* **29**(4), 297-315.

Hickin, E.J. (1984) Vegetation and river channel dynamics. *The Canadian Geographer,* **38**(2), 111-126.

Hupp, C.R. (1982) Stream-grade variations and riparian-forest ecology along the Passage Creek, Virginia. *Bulletin of the Torrey Botanical Club,* **109**, 488-499.

Hupp, C.R. (1983) Vegetation patterns on channel features in the Passage Creek, Virginia. *Castanea,* **48**, 62-72.

Hupp, C.R. and Bazemore, D.E. (1993) Temporal and spatial patterns of wetland sedimentation, West Tennessee. *Journal of Hydrology,* **141**, 179-196.

Hupp, C.R. and Morris, E.E. (1990) A dendrogeomorphic approach to measurement of sedimentation in a forested wetland, Black Swamp, Arkansas. *Wetlands,* **10**, 107-124.

Hupp, C.R. and Osterkamp, W.R. (1985) Bottomland vegetation distribution along Passage Creek, Virginia in relation to fluvial landforms. *Ecology,* **66**, 670-681.

James, C.S. (1985) Sediment transfer to overbank sections. *Journal of Hydraulic Research,* **23**, 435-452.

James, C.S., Nicolson, C.R., Van Niekerk, A.W. and Heritage, G.E. (1996) *Fluvial Geomorphology and Fish Habitat: Implications for River Restoration.* 2nd international symposium on habitat hydraulics, Québec, INRS-Eau & IAHR.

Johnson, W.C., Burgess, R.L. and Keammerer, W.R. (1976) Forest overstory vegetation and environment on the Missouri river floodplain in North Dakota. *Ecological Monographs,* **46**, 59-84.

Junk, W.J. and Welcomme, R.L. (1990) Floodplains. In: Patten, B.C. (Ed.) *Wetlands and shallow continental Waterbodies.* The Hague, SPB Academic publishing bv. pp 491-524.

Kellerhals, R. and Miles, M. (1996) *Fluvial Geomorphology and Fish Habitat: Implications for River Restoration.* 2nd international symposium on habitat hydraulics, Québec, INRS-Eau & IAHR.

Kelsey, H.M., Lamberson, R. and Madej, M.A. (1987) Stochastic model for the long-term transport of stored sediment in a river channel. *Water Resource Research,* **23**, 1738-1750.

Knighton, A.D. (1984) *Fluvial forms and processes.* London, Edward Arnold.

Kozlowski, T.T. (1984) Responses of woody plants to flooding. In: Kozlowski, T.T. (Ed.) *Flooding and Plant Growth.* Orlando, Academic Press. pp 129-163.

Langlade, L.R. and Décamps, O. (1994) Plant colonisation on river gravel bars: the effect of litter accumulation. Comptes Rendus de l'Académie des Sciences (Paris), *Sciences de la Vie,* **317**, 899-905.

Latulippe, C. and Peiry, J.-L. (1996) Essai de hiérarchisation des zones de production de matière en suspension dans le bassin versant d'un grand cours d'eau: l'Isère en amont de Grenoble. *Revue de Géographie Alpine,* **84**(2), 29-44

Lavertu, D., Mauffette, Y. and Bergeron, Y. (1994) Effects of stand age and litter removal on the regeneration of *Populus deltoides. Journal of Vegetation Science,* **5**, 561-568.

Leuthold, B. (1994) Vegetations- und Standortsveränderungen auf von Hochwasser überschlickten Streuwiesen. *Veröffentichungen des geobotanischen Institutes der Eidg. tech. Hochschule Stiftung Rübel in Zürich,* **121**, 1-121.

Lowrance, R., Sharpe, J.K. and Sheridan, J.M. (1986) Long-term sediment deposition in the riparian zone of a coastal plain watershed. *Journal of Soil and Water Conservation,* **41**, 246-271.

Malanson, G.P. (1993) *Riparian Landscapes.* Cambridge, Cambridge University Press.

Malanson, G.P. and Butler, D.R. (1990) Woody debris, sediment and riparian vegetation of a subalpine river, Montana, USA. *Arctic and Alpine Research,* **22**, 183-194.

Marston, R.A. (1982) The geomorphic significance of log steps in forest streams. *Annals of the Association of American Geographers,* **72**, 99-108.

Marti, E. and Sabater, F. (1996) High variability in temporal and spatial nutrient retention in Mediterranean streams. *Ecology,* **77**(3), 854-869.

Meade, R. and Blackstock, T.H. (1988) The impact of drainage on the distribution of rich-fen plant communities in two Anglesey Basins. *Wetlands,* **8**, 159-177.

Mitsch, W.J. and Gosselink, J.G. (1993) *Wetlands.* New York, Van Nostrand and Reinhold.

Montgomery, D.R., Abbe, T.B., Buffington, J.M., Peterson, N.P., Schmidt, K.M. and Stock, J.D. (1996) Distribution of bedrock and alluvial channels in forested mountain drainage basins. *Nature,* **381** (13 June 1996), 587-589.

Moore, P.D. and Evans, A. (1991) The development of valley mires in South East England. *Aquilo Ser. Bot.,* **30**(1), 25-34.

Morin, E., Bouchard, A. and Jutras, P. (1989) Ecological analysis of disturbed river banks in the Montreal area of Québec. *Environmental Management,* **13**, 215-225.

Morizawa, M. (1985) *Rivers.* London, Longman.

Murray, B.A. and Paola, C. (1994) A cellular model of braided rivers. *Nature,* **371** (1 September 1994), 54-57.

Nanson, G.C. and Beach, H.F. (1977) Forest succession and sedimentation on a meandering river floodplain, Northeast British Columbia, *Canada. Journal of Biogeography,* **4**, 229-251.

Nilsson, C. and Grelsson, G. (1990) The effects of litter displacement on river bank vegetation. *Canadian Journal of Botany,* **68**, 735-741.

Pautou, G., Thomas, J., Aïn, G., Meriaudeau, R., Gilot, B. and Neuburger, M.-C. (1972) *Etude écologique des îles du Rhône entre Seyssel et Lyon.* Grenoble, Université de Grenoble.

Pautou, G., Girel, J., Aïn, G. and Lachet, B. (1979) Recherches écologiques dans la vallée du Haut-Rhône Français. *Documents de Cartographie Ecologique,* **22**, 5-63, 2 cartes dépliantes hors texte.

Pautou, G. (1980) La dynamique de la végétation dans la vallée du Rhône entre Genève et Lyon. *Colloques phytosociologiques,* **9**, 81-91.

Pautou, G. (1984) L'organisation des forêts alluviales dans l'axe rhodanien entre Genève et Lyon; comparaison avec d'autres systèmes fluviaux. *Documents de Cartographie Ecologique,* **27**, 43-64.

Pautou, G. and Décamps, H. (1985) Ecological interactions between the alluvial forests and hydrology of the upper Rhône. *Archiv. Hydrobiol.,* **104**, 13-37.

Pautou, G., Girel, J. and Maman, L. (1985) Le rôle des processus allogéniques dans le déroulement des successions végétales, l'exemple de la plaine du Rhône entre Genève et Lyon. *Colloques phytosociologiques,* **13**, 655-668.

Pautou, G. and Wuillot, J. (1989) La diversité spatiale des forêts alluviales dans les îles du Haut-Rhône Français. *Bulletin d'Ecologie,* **20**(3), 211-230.

Pautou, G., Girel, J., Pautou, M.-P. and Gruffaz, R. (1995) Hydrologie, paludisme et démoustication (l'exemple de la région Rhône-Alpes), *Revue de géographie alpine (Grenoble),* **83**(1), 33-52.

Pautou, G., Girel, J., Peiry, J.L., Hughes, F., Richards, K., Foussadier, R., Garguet-Duport, B., Harris, T. and Barsoum, N. (1996) Les changements de végétation dans les hydrosystèmes fluviaux; l'exemple du Haut-Rhône et de l'Isère dans le Grésivaudan. *Revue d'Ecologie Alpine,* **3**, 41-66.

Peterken, G.F. and Hughes, F.M.R. (1995) Restoration of floodplain forests in Britain. *Forestry,* **68**(3), 187-202.
Petts, G. and Foster, I. (1985) *Rivers and Landscape.* London, Edward Arnold.
Phillips, J.D. (1989) Fluvial sediment storage in wetlands. *Water Resources Bulletin,* **25**, 867-873.
Schopp-Guth, A., Maas, D. and Pfadenhauer, J. (1994) Influence of management on the seed production and seed bank of calcareous fen species. *Journal of Vegetation Science,* **5**, 569-578.
Sparks, R.E., Bayley, P.B., Kohler, S.L. and Osborne, L.L. (1990) Disturbance and recovery of large floodplain rivers. *Environmental Management,* **14**, 699-709.
Steven, L.E., Schmidt, J.C., Ayers, T.J. and Brown, B.T. (1995) Flow regulation, Geomorphology and Colorado River Marsh Development in the Grand Canyon, Arizona. *Ecological Applications*, **5**(4), 1025-1039.
Swanson, F.J., Gregory, S.V., Sedell, J.R. and Campbell, A.G. (1982) Land-Water Interactions: The Riparian Zone. In: Edmonds, R.L. (Ed.) *Analysis of coniferous forest ecosystems in the Western United States,* US/IBP Synthesis, 14. Stroudsburg, Hutchinson Ross Publishing. pp 267-291.
Tsujimoto, T., Kitamura, T. and Murakami, S. (1996) *Basic Morphological Processes due to Deposition of Suspended Sediment affected by Vegetation.* 2nd international symposium on habitat hydraulics, Québec, INRS-Eau & IAHR.
Verhoeven, J.T.A. and Van der Thoorn, J. (1990) Wetland management; Marsh eutrophication and waste water treatment. In: Patten, B. (Ed.) *Wetlands and Shallow Continental Waterbodies.* The Hague, SPB Academic publish. pp 571-585.
Waldemarson-Jensen, E. (1979) Succession in relationship to lagoon development in the Laitaure delta, North Sweden. *Acta Phytogeographica Suecica,* **66**, 1-128.
Walker, L.R. and Chapin, F.S. (1986) Physiological controls over seedling growth in primary succession on an Alaskan floodplain. *Ecology,* **67**, 1508-1523.
Zimmerer, K.S. (1995) The origins of Andean irrigation. *Nature,* **378** (30 November 1995). 481-483.

Part II

The Potential of Habitat Types to Act as Buffer Zones

The Potential Role of In–Stream and Hyporheic Environments as Buffer Zones

12

A.R. Hill

Department of Geography, York University, 4700 Keele Street, North York, Ontario, M3J 1P3, Canada

Abstract

Concern about increased nitrogen (N) and phosphorous (P) inputs to streams has stimulated research on processes that regulate stream nutrient transport. Most research has focused on in-stream processes involving interactions at the sediment-water interface. However, nutrient transformations also occur in the hyporheic (deep sediment) zone. Denitrification in biofilms and surface sediments has frequently been identified as the major mechanism of nitrate removal from stream water. The main limiting factor for denitrification is the rate at which nitrate is supplied to denitrifying sites. Other factors influencing denitrification include oxygen, temperature and organic matter concentrations. Whereas N retention is controlled mainly by biological processes, geochemical sorption reactions with stream sediments often play a major role in P retention. Limited research indicates that various nutrient transformations including nitrification and denitrification occur in hyporheic zones and influence bidirectional nutrient exchanges between the stream and the subsurface environment. Research on hyporheic zones has been "stream centred" rather than focused on groundwater interactions. Knowledge is lacking on the role of the hyporheic environment as a buffer for nitrate rich groundwater which flows at depth beneath riparian zones and discharges directly through the streambed. Mass balance studies suggest that streams frequently remove <10% of annual N and P inputs, whereas retention often ranges from 20-80% of inputs during low summer flows. The capacity of streams to retain N and P is limited mainly by short water residence times associated with periods of high stream discharge when most N and P is transported. Although in-stream and hyporheic environments are probably not an important buffer in relation to annual N and P fluxes, effective seasonal retention alters the timing of transport and may influence the dynamics of downstream ecosystems. Management of stream habitats to increase water residence times and the accumulation of fine organic-rich sediments in stream channels could enhance the efficiency of N and P retention, particularly during periods of low flow.

INTRODUCTION

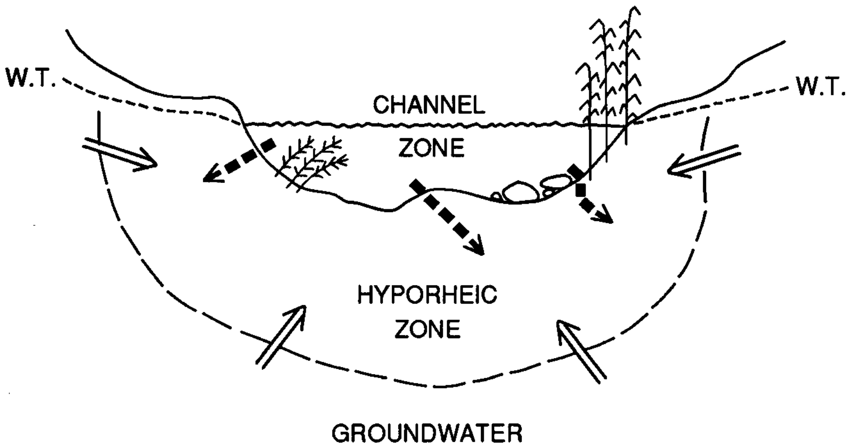
In recent years upward trends in nitrogen and phosphorus levels in streams arising from point and diffuse source inputs have produced concern because of ecological impacts on both freshwater and estuarine ecosystems (Heathwaite *et al.*, 1996). This concern has stimulated research on processes that regulate the transport and transformation of nutrients in streams and rivers. In the past two decades an earlier perspective of streams as relatively inert pipelines which transport nutrients from catchments to lakes and oceans has been replaced with a focus on streams as dynamic ecosystems which continually exchange water and solutes with stream beds and the adjacent stream valley environment (Bencala, 1993). These exchanges which have a potential to influence the chemical and physical forms of nutrients occur within the framework of continual downstream transport. This coupling of cycling and downstream movement is expressed in the nutrient spiralling concept (Newbold *et al.*, 1981; Newbold, 1994).

Research on stream nutrient transport and transformations has focused mainly on in-stream processes that involve interactions between the stream bed surface and stream water (Fig. 1). However, since the late 1980's stream ecologists have also begun to examine nutrient transformations within the stream

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0 © 1997 Quest Environmental.

hyporheic zone; a subsurface region where mixing of streamwater and groundwater occur adjacent to the stream channel (Triska *et al.,* 1989a; Valett *et al.,* 1993). The recent recognition of the hyporheic zone has expanded the spatial extent of the stream ecosystem to include a vertical and horizontal dimension around the channel.

Figure 1. The channel and hyporheic zone of a stream ecosystem.



The objective of this paper is to review the current state of knowledge on the role of in-stream and hyporheic environments as buffer zones that regulate the stream transport of nitrogen (N) and phosphorus (P). Emphasis will be placed mainly on nutrient-rich lowland streams in temperate landscapes since this is where most research has been done. The role of large riverine wetlands as buffer zones is considered by Johnston *et al.* in a related paper in this volume. Attention is centred on stream reaches with running water. Small ponds often occur along streams in agricultural landscapes and their potential role as buffers is assessed by Fleischer *et al.* in a later chapter.

IN–STREAM N AND P REMOVAL

Although N occurs in streams as organic N, ammonium (NH_4^+) and nitrite (NO_2^-), the predominant form of N in lowland agricultural streams is usually nitrate (NO_3^-). Research has focused on nitrate transport and transformations in streams because of its effects on water quality and health concerns over high nitrate concentrations in drinking water (Fraser and Chilvers, 1981). Studies of stream P transport have commonly examined soluble reactive phosphorus (SRP) and particulate phosphates (PP) adsorbed to sediment particles. The analytical method used in the measurement of SRP includes easily hydrolysed organic phosphorus compounds in addition to dissolved inorganic phosphates (Stainton, 1980). Although SRP often comprises a small proportion of the total annual P flux in streams, it is the most bioavailable P form.

Mechanisms for nitrate removal

Several processes are involved in the removal of nitrate from stream water. These include uptake by aquatic macrophytes and benthic algae for growth. Dissimilatory reduction of nitrate by biological denitrification occurs under anaerobic conditions. This microbial process involves the reduction of NO_3^- to NO_2^- which is further reduced to gaseous N forms (NO, N_2O N_2) that are lost to the atmosphere (Tiedje, 1982).

Several studies suggest that uptake by aquatic macrophytes is an important nitrate depletion process in some streams. In the River Raan in southern Sweden macrophyte vegetation dominated by *Glyceria maxima and Potamogeton* spp. with dense stands of attached algae removed 40-70% of NO_3-N input during summer (Jansson *et al.,* 1994). In a New Zealand stream where the channel was vegetated with the semi-aquatic sweet floating grass *(Glyceria fluitans)* 75% of nitrate removal was attributed to plant uptake (Cooper and Cooke, 1984). In a second New Zealand stream watercress *(Nasturtium officinale)* accounted for 75% of the nitrogen loss during late summer (Vincent and Downes, 1980). Detailed field measurements in this stream showed that watercress accumulated 1.14g N m^{-2} d^{-1} in mid-summer and this uptake could account for all the nitrate loss of 344 kg N in the stream reach (Howard-Williams *et al.*, 1982, 1983).

In contrast, other studies indicate that plant uptake is not important in stream nitrate removal. Macrophytes, mostly watercress, accounted for about 5% of the nitrogen loss in a small spring-fed Ontario stream (Kaushik *et al.*, 1975). Macrophytes also removed less than 2% of annual NO_3-N export in several English chalk streams (Casey, 1977) and were not considered important in two small streams near Oslo, Norway (Faafeng and Roseth, 1993). Little information is available on the role of algae in nitrate uptake from stream water. Nitrate removal during stream enrichment experiments has been attributed to uptake by benthic algae (McColl, 1974; Sebetich *et al.,* 1984) and nitrate uptake was correlated with algae abundance and net production in several desert streams in Arizona (Grimm *et al.*, 1981).

Although uptake by aquatic plants may play a role in regulating nitrate transport, this mechanism does not permanently remove nitrate from the stream. Much of the N removed by aquatic plants is recycled to the stream in decomposing plant tissues. Howard-Williams *et al.* (1983) calculated that the nitrogen uptake by watercress in a New Zealand stream was released during decomposition as nitrate (56%), particulate organic nitrogen (23%) and refractory dissolved organic nitrogen (21%).

Denitrification activity has been measured in stream sediments by the acetylene inhibition technique. Most studies have been conducted in relatively nutrient-rich streams with denitrification rates generally ranging from 10-300 mg N m^{-2} d^{-1} (Table 1). Laboratory incubation of sediment cores suggests that nitrate is depleted rapidly below the sediment-water interface in the upper few cms of most sediments (Sain *et al.*, 1977; Hill, 1981; Wyer and Hill, 1984). Measurements of Eh indicate that the oxidising zone is often less than 1 cm thick (Sain *et al.*, 1977; Cooke and White, 1987).

Table 1. Denitrification rates measured in stream sediments

Location	*Stream NO_3-N (mg L^{-1})*	*Denitrification rate (mg N m^{-2} d^{-1})*	*Reference*
Swifts' Brook Canada	9.1*	50-90	Chatarpaul *et al.*, 1980
Nottawasaga River Canada	2.0*	10-190	Hill, 1983b
Purukohukohu New Zealand			Cooper and Cook, 1985
forest	0.05-0.5	4.8-122	
pasture	0.4-2.0	122-492	
Duffin Creek Canada	5.0*	50-380	Hill and Sanmugadas, 1984
River Dorn, England	8.6	34-773	Cooke and White, 1987
Scotsman Valley New Zealand	0.6-4.1	0.3-5.6	Cooke and Cooper, 1988
Gelbaek Denmark	0.05-0.5	14-126	Christensen and Sorensen, 1988
Rabis Baek Denmark			Christensen and Sorensen, 1988
unvegetated	0.1-0.15	14-56	
vegetated	0.1-0.15	14-154	
Arhus Å Denmark	0.11	114	Christensen *et al.*, 1989
Gudenå Denmark	0.55	333	Christensen *et al.*, 1989

* laboratory nitrate solutions

The recent development of micro-electrodes has facilitated a more detailed analysis of denitrification in sediments. Christensen *et al.* (1989) showed that denitrification started at 0.2 mm below the sediment surface and extended over a 3.6 mm thick zone in sediments from Gardena stream in Denmark, whereas in a second stream it began approximately 0.7 mm below the surface and was restricted to a 0.7 mm sediment thickness. In both streams denitrification occurred mainly just below the oxic-anoxic interface where nitrate was present as a result of diffusion from overlying stream water. Measurements of N_2O and O_2 indicate that the diffusional supply of nitrate from overlying water is controlled primarily by the thickness of the oxic layer (Christensen *et al.*, 1989).

Considerable temporal and spatial variations in denitrification rates occur in stream sediments. Maximum rates are frequently observed in spring and summer with winter rates often being only 10% of summer values (Jansson *et al.*, 1994). Although nitrate removal has been measured in stream sediments which range from acidic sands to calcareous fine grained sediments, denitrification rates are often lower in coarse sands and gravels in comparison to fine silty sediments (Hill, 1983a; Wyer and Hill, 1984). Denitrification is not confined to stream sediments. It has also been measured in various periphyton communities ranging from algae mats to slime enmeshed microbial communities on rock and gravel surfaces (Triska and Oremland, 1981; Ventullo and Rowe, 1982; Duff *et al.*, 1984).

The availability of nitrate frequently controls the rate of denitrification in stream sediments. In organic-rich sediments from a Danish stream, denitrification was stimulated 5-20 X by nitrate amendments (Christensen and Sorensen, 1988). Nitrate also stimulated denitrification in sediments with 1-15% organic carbon from a Swedish stream (Jansson *et al.*, 1994). Denitrification in stream sediments also depends upon the availability of organic carbon as an energy source. Hill and Sanmugadas (1985) reported a strong relationship between nitrate loss from laboratory incubated sediment cores and the water soluble carbon content of 0-5 cm sediments from three Ontario streams. A strong correlation between *in situ* denitrification activity and readily mineralizable carbon was also found in sediments from the River Don in England (Cooke and White, 1987). Addition of carbon (acetate, glucose) stimulated denitrification activity in sandy sediments with <1% organic carbon (Jansson *et al.*, 1994). However, a transition from carbon limitation to nitrate limitation occurred when sediment organic carbon exceeded 1%. These data suggest that the principal limiting factor for denitrification is the rate at which nitrate is supplied even in sediments with relatively low organic contents of 1-2%.

Denitrification is also affected by temperature and studies of sediment cores have shown that the rate of denitrification increased with increasing temperature in the range of 5-23°C (Sain *et al.*, 1977). Nitrate removal by stream sediments has been measured at temperatures as low as 2°C although rates of loss were only 20% of losses at 20°C (Hill, 1983a). Animals can enhance the rate of denitrification in stream sediments. The use of ^{15}N as a tracer indicated that when tubificid worms were present in sediments the rate of denitrification was increased by 80% from 50 to 90 mg N m^{-2} d^{-1} (Chatarpaul *et al.*, 1980). Bioturbation increases the diffusion of nitrate and oxygen to deeper sediment zones and also increases oxygen for nitrification in surface sediments thus increasing overall rates of coupled nitrification/denitrification within the sediments (Henriksen *et al.*, 1983). The presence of macrophytes in streams may also influence denitrification activity. Macrophytes can increase sediment organic content by trapping organic detritus and releasing DOC from roots (Christensen and Sorensen, 1986). The release of oxygen from roots surfaces may also stimulate nitrification thus providing a source of nitrate for denitrification (Christensen and Sorensen, 1988).

Mechanisms for P removal

In contrast to nitrate, retention of inorganic P in streams is strongly influenced by abiotic processes. Soluble reactive phosphorus (SRP) interacts strongly with suspended and bed sediments by means of adsorption and desorption reactions (McCallister and Logan, 1978; Klotz, 1985). In hardwater streams inorganic P can also form coprecipitates with calcite (House, 1990). Biological uptake of P by plants, algae and micro organisms also influence P transformations and transport in streams (Elwood *et al.*, 1981).

Biological cycling of P has been studied in a considerable number of small undisturbed headwater streams (Meyer, 1979; Mulholland *et al.*, 1985; Munn and Meyer, 1990), but limited data are available on P retention by macrophytes and algae in eutrophic lowland streams. Submerged macrophytes in a Danish stream were estimated to remove 6-24 kg P from stream water and sediments which represented only 0.5-2% of the SRP load (Svendsen and Kronvang 1993). In the same stream, benthic algae biomass data indicated a SRP uptake of approximately 54 kg P. This estimate does not consider the rapid turnover rate of algae so that total uptake may be several times higher (Svendsen *et al.*, 1995). Casey and Clark (1986) estimated that P uptake by algae was equivalent to 50% of SRP transport in April suggesting that this mechanism of P removal can be seasonally important in some streams. Phosphorus uptake by macrophytes and algae is released back to the stream after decomposition mainly as particulate organic P and may be exported before conversion to SRP or dissolved organic P if water residence times are short (Svendsen *et al.*, 1995).

P adsorption and desorption processes in streams have been widely studied (Green *et al.*, 1978; Klotz, 1985; Stone and Mudroch, 1989; House *et al.*, 1995). This research indicates that sediments act as an important buffering mechanism which largely controls the dissolved-particulate P balance in most relatively eutrophic lowland streams. Sediment-water interactions also regulate the biological availability and transport of P in streams (Dorioz *et al.*, 1989; Klotz, 1991; Stone *et al.*, 1991). The stream water dissolved P concentration often coincides with the EPC_o of sediments suggesting that the amount of SRP in streams is regulated by sediment interactions (Taylor and Kunishi, 1971; Meyer, 1979). The zero equilibrium phosphate concentration (EPC_o) is the phosphate concentration in solution at which there is neither adsorption nor desorption of P by sediments. Klotz (1988) found a significant positive correlation between downstream increase in stream SRP concentrations and the EPC_o of sediments in a small New York stream.

The phosphate sorption index which provides a useful measure of the phosphate buffering capacity of sediment (Bache and Williams, 1971) has been analysed in sediments from several streams. Large amounts of P can be adsorbed by sediments with high index values without increasing the EPC_o of stream water. Phosphate sorption indices of 8.3 and 12.3 for two agricultural streams in New York indicated a greater potential to sorb P than two adjacent forest streams with indices of 2.45 and 1.89 (Klotz, 1985). Two P rich streams in southern Ontario had high indices of 14.2 for silty sediments in Duffin Creek and 8.6 for sandy low organic carbon sediments in the Nottawasaga river (Hill, 1982). Meyer (1979) found that silty and sandy sediments from Bear Brook, a forested upland stream, had an index of 10.3 and 2.1 respectively.

The sorption of P is influenced by chemical and physical properties of stream sediments. Increased sorption is associated with silt and clay fractions and organic matter content (Stone and Mudroch, 1989; Stone *et al.*, 1995). Several studies indicate strong positive correlations of P sorption with amorphous Fe and Al oxides and hydroxides in sediments (Shulka *et al.*, 1971; Green *et al.*, 1978; McCallister and Logan, 1978). Svendsen *et al.* (1995) calculated that 50-75% of sediment P pool was Fe-P in a Danish stream suggesting that SRP removal from stream water was caused in part by adsorption to Fe and Al hydroxides.

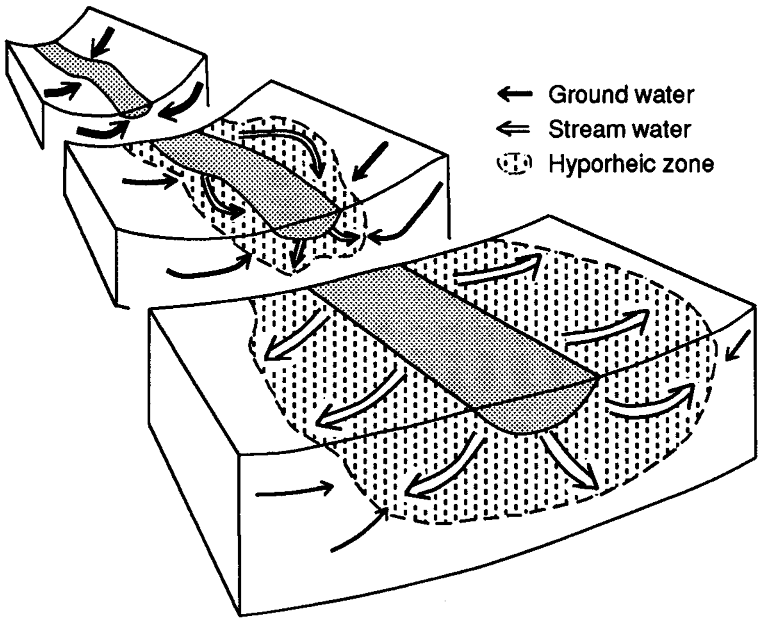
ROLE OF THE HYPORHEIC ZONE IN N AND P REMOVAL

In the past decade research has begun to focus on the delineation of stream hyporheic zones and the analysis of element transformations within this subsurface environment. Exchanges of stream and groundwater in the hyporheic zone influence stream chemistry as a result of storage and retention as well as biogeochemical processes (Valett *et al.*, 1994). The hyporheic zone may be a sink or source of solutes to the stream depending on the relative importance of processes which immobilise or generate nutrients in the sediments (Triska *et al.*, 1989a).

At a catchment scale, the lateral and vertical extent of hyporheic zones is controlled by landscape properties such as geological lithology, groundwater magnitude and seasonality, hillslope and channel gradients and annual precipitation (Valett *et al.*, 1996). At the individual stream reach scale the size of the hyporheic zone is determined by geomorphological features of the surface channel such as permeability and variations in stream bed topography which influence stream water slope. Features such as boulders, gravel bars and riffle-pool sequences produce rapid changes in elevation which force streamwater into the subsurface producing localised flow paths in the sediments (Thibodeaux and Boyle, 1987; Harvey and Bencala, 1993).

The size of the hyporheic zone and the extent of interaction between stream water and groundwater may vary in a downstream direction from headwaters to large rivers (Fig. 2). Small perennial headwater streams often have small hyporheic zones on the scale of centimetres because of large groundwater fluxes (White, 1993). The hyporheic zone associated with pool-riffle sequences in mid-order streams can have a vertical and lateral extent on the scale of metres. Tracers injected into a third order gravel bed stream in northern California revealed a high percent of stream water at lateral distances of up to 10 m from the channel (Triska *et al.*, 1993). Large scale hydrological exchanges of surface and groundwater extending over distances of kilometres have been described for large river floodplains (Stanford and Ward, 1993). Recently, conceptual models have been proposed which suggest that the importance of the hyporheic zone to stream chemistry is influenced by the proportion of stream water passing through this zone, as well as by water residence time and the rates of biogeochemical processes within the hyporheic zone (Findlay, 1995; Valett *et al.*, 1996).

Figure 2. The hypothetical representation of trends in the size of the hyporheic zone, from headwaters to large rivers.



Several studies have shown the occurrence of nitrification and denitrification in stream hyporheic zones during summer base flows (Duff and Triska, 1990; Triska *et al.*, 1993; Jones *et al.*, 1995; Valett *et al.*, 1996). Nitrogen injection experiments indicated that ammonium in groundwater entering the hyporheic zone of a northern California stream was oxidised to nitrate in aerobic areas of the zone, whereas nitrate transported to low dissolved oxygen regions was either denitrified or reduced to ammonium (Triska *et al.*, 1993). The net effect of various N cycling processes in the hyporheic zone of this stream was an overall increase in nitrate concentration of stream water (Triska *et al.*, 1989b). Nitrate concentrations in stream water flowing into the hyporheic zone of a desert stream were also elevated by nitrification (Jones *et al.*, 1995). Nitrate rich subsurface water was an important source of N for stream algae as the water re-entered the stream (Valett *et al.*, 1994). Analysis of subsurface flow through a 200 m long gravel bar on the River Garonne indicated rapid NO_3-N depletion in the first few metres of the bar from 2-4 mg L^{-1} in the stream to <1 mg L^{-1} in the bar (Pinay *et al.*, 1994). Measurements of *in situ* denitrification indicated high rates at the upstream end of the bar in areas of organic rich silt and a sharp decline in a downstream direction where sediments were sandy.

Little information is available on P transformations in stream hyporheic zones. Several studies noted that SRP concentrations were often higher in the hyporheic zone than in surface water (Valett *et al.*, 1990; Hendricks and White, 1995). The hyporheic zone may be an important source of SRP to streams during periods of surface water nutrient depletion. In a riffle-pool sequence of a third order sand bottom river in northern Michigan, SRP was significantly enriched relative to stream water particularly beneath the downstream end of the riffle where upwelling occurred (Hendricks and White, 1995). White *et al.* (1992) noted that macrophytes were often located in these upwelling areas where SRP was enriched.

Despite these recent studies of particular stream hyporheic zones, knowledge of the role of this subsurface environment as a nutrient buffer zone is limited. Most research has focused on streams with low nutrient loadings in forest and desert landscapes. In these streams the hyporheic zone is frequently a nutrient source for surface waters rather than a sink. It is possible that the hyporheic zone of eutrophic streams may remove nutrients from stream water. However, research on stream denitrification and P adsorption suggest that these processes occur mainly near the sediment-water interface. Consequently, considerable portions of the hyporheic zone may not be involved in the depletion of nutrients in stream water.

Previous research has focused on how stream water is modified by interaction with the hyporheic zone. This stream centred perspective has resulted in an absence of research on the regulation of groundwater nutrient fluxes within this zone. In landscapes where shallow impermeable layers are absent, large groundwater inputs from adjacent uplands may flow at depth beneath stream riparian zones and discharge directly through the stream bed (Phillips *et al.*, 1993; Bohlke and Denver, 1995; Hill, 1996). Little information is available on the capacity of stream hyporheic zones to buffer large nitrate fluxes in groundwater inputs. Robertson *et al.* (1991) noted some temporal variability in the removal of nitrate in septic tank effluent that flowed upward through the bed of a small Ontario stream. Although NO_3-N concentrations generally declined from 20 mg L^{-1} to <0.5 mg L^{-1} in the last metre of the flow path, in July concentrations rose briefly to 13 mg L^{-1} at the bed surface before returning to <0.5 mg L^{-1}. In a Maryland agricultural catchment, nitrate contaminated groundwater flowed at depth in a thick sand aquifer beneath the riparian zone and discharged upward through the stream bed (Bohlke and Denver, 1995). Although no detailed data are available for the hyporheic zone of this stream, the high stream base flow NO_3-N concentration of 7-10 mg L^{-1} suggests that the stream bed was not an effective nitrate buffer.

THE IMPORTANCE OF N AND P REMOVAL IN STREAMS

The magnitude of N and P removal from streams in relation to seasonal and annual fluxes has been examined by mass-balance calculations in a range of lowland nutrient-rich streams. This analysis usually involves the measurement of N and P loads of a stream entering and exiting a particular reach. Additional measured inputs to the reach may include small tributary streams and groundwater flow. In a few studies the magnitude of individual removal processes, such as denitrification or P adsorption by sediments has been measured directly and compared to calculations of seasonal or annual nutrient fluxes.

Estimates of the magnitude of nitrate removal during the summer season when temperatures are warm and streams are frequently at base flow range from <10% to 76% (Table 2). The very high removal rates of 60 and 76% of NO_3-N flux in Swifts Brook and Duffin Creek respectively are mean rates calculated for a number of individual observation days during stream low flows in late May-October. In contrast, the lower NO_3-N removal estimates of 10% and 0-13% in a Danish and Swedish stream (Table 2) are calculated for the entire summer and include occasional periods of high stream flow. Analysis of the River Raan in southern Sweden illustrates the importance of summer storm flow events to overall nitrate retention during this season (Jansson *et al.*, 1994). Nitrate -N removal was 0% in the summer of 1987 which was cold with higher than normal runoff and 13% in 1988, a dry summer with low runoff. Annual mass-balances indicate that NO_3-N removal ranges from 1-5% in many rivers, although higher removal rates of 17-22% and 68% were estimated in Swift's Brook, Ontario and the Puruki stream in New Zealand respectively (Hoare, 1979; Robinson *et al.*, 1979).

Table 2. Nitrate removal in lowland streams in agricultural catchments.

Location	*-% of N loading-* *Summer low flow*	*Annual*	*Reference*
Swifts' Brook Ontario	60% [(a)]	17-22% [(b)]	Kaushik and Robinson 1976; Robinson *et al.*, 1979
Duffin Creek Ontario	76% [(a)] 48% [(b)]	5-6% [(b)]	Hill, 1979; 1981
Puruki Stream New Zealand		68% [(a)]	Hoare, 1979
Nottawasaga River Ontario	13% [(a)] 11% [(b)]	1% [(b)]	Hill, 1983a
Dorn River England	15% [(c)]		Cooke and White, 1987
Gelbaek and Rabis Baek Denmark	10% [(c)]	1% [(c)]	Christensen and Sorensen, 1988
Råån Stream Sweden	0-13% (b)	<2% (b)	Jansson *et al.*, 1994

[(a)] NO_3-N loading
[(b)] total N loading
[(c)] direct denitrification measurements

The critical factor in explaining these contrasts between daily, seasonal and annual nitrate retention is the residence time of water (Hill, 1988; Jansson *et al.*, 1994). Significant removal of NO_3^- from stream water only occurred in a 7 km reach of the River Raan, Sweden during summer low flows when the water residence time was 1-2 days (Jansson *et al.*, 1994). Nitrate -N removal as a % of daily input declined rapidly from maximum values of 30-75% at stream discharges of 0.4 m^{-3} sec^{-1} to <1% at 1.4 m^{-3} sec^{-1} in three reaches of Duffin Creek (Hill 1988). A similar pattern of declining nitrate removal efficiency as discharge increased occurred in the R. Raan (Jansson *et al.*, 1994).

The effect of stream discharge on nitrate removal efficiency is not only linked to reduced residence times as water velocity increases, but also to larger nitrate fluxes (Cooper, 1990). An increase in N inputs associated with high discharge produces a decline in removal efficiency even if nitrate removal rates remain constant or increase by an amount which represents a lower proportion of the N input. In temperate streams most N transport occurs during periods of high stream discharge associated with low water temperatures in the winter months (Hill, 1986; Jansson *et al.*, 1994). This combination of conditions severely limits the effectiveness of nitrate removal mechanisms. Consequently, the proportion of seasonal and annual NO_3-N load transported by high discharge exerts a major influence on the importance of nitrate removal in relation to N fluxes. Stormflow represents a high proportion of annual runoff in catchments with thin soils and impermeable parent materials, whereas base flows often contribute most of the annual runoff in catchments with high infiltration rates and extensive aquifers (Walling, 1971). It is noteworthy that the two streams, Swift's Brook and Puruki, shown in Table 2, which have a very large annual percent removal of nitrate, are small groundwater fed systems. Stormflow accounted for only 10% of the annual runoff in Puruki stream (Hoare, 1979).

Phosphorus removal during summer periods ranges from 20-92% in relatively eutrophic streams (Table 3). This considerable range reflects in part differences in observation periods and calculation methods. The high P removal rates of 44 and 92% in the Nottawsaga river and Duffin Creek were estimated from SRP mass-balances for individual summer days of low stream flow (Hill, 1982). In the Redon river, total P mass balances were calculated for 4 separate weeks in summer and autumn without rain when stream discharge was <0.2 m^{-3} sec^{-1} (Dorioz *et al.*, 1989, Table 8). The lower P removal % in two Danish streams (Gelback and Gjern Å) is based on *in situ* stream bed measurements of P retention in comparison to total TP export for the entire summer period which included storm flow events (Svendsen *et al.*, 1995). It should be noted that the relatively high summer P retentions shown in Table 3 were measured in streams where almost all the P input is delivered by point sewage sources during low flow periods. It is uncertain whether similar large accumulations of stream bed P would occur in streams which only receive diffuse P inputs from cropland.

Table 3. Phosphorus removal in lowland streams in agricultural catchments.

Location	*-% of P loading-* *Summer low flow*	*Annual*	*Reference*
Duffin Creek Ontario	92% [a]		Hill, 1982
Nottawasaga River Ontario	44% [a]		Hill, 1982
Redon River France	51% [b]		Dorioz *et al.*, 1989
Gelbaek Denmark			Svendsen *et al.*, 1995
1987	36% [c]		
1988	65% [c]		
Gjern Å Denmark			Svendsen *et al.*, 1995
1987	33% [c]	56% [a] <5% [b]	
1988	20% [c]	59% [a]	

[a] SRP loading
[b] total P loading
[c] total P removal calculated as % of net TP export

Detailed measurements of P retention in streams have usually been limited to summer months. However, a mass balance study for Gjern Å in Denmark revealed that the river was a significant sink for SRP during all seasons except for a high flow period in the winter of the first observation year (Svendsen *et al.*, 1995). The net retention of SRP was 1220 kg P in 1987-88 and 1600 kg P in 1988-89, equivalent to 56 and 59% respectively of the annual SRP export (Table 3). Although these data suggest significant retention of SRP throughout the year, particulate phosphorus (PP) is usually the dominant form of P accounting for 56-80% of total P export in lowland streams (Johnson *et al.*, 1976; Svendsen *et al.*, 1995). Annual retention of TP in Gjern Å was negligible compared with P export (Svendsen *et al.*, 1995).

The export of P accumulated in stream bed sediments during storm flows is a major reason for the absence of significant annual retention of total P in streams (Harms *et al.*, 1978; Dorioz *et al.*, 1989). Storm flows occurring after considerable periods of low flows have very high concentrations of PP (Johnson *et al.*, 1976; Dorioz *et al.*, 1989; Svendsen and Kronvang, 1993). Increased SRP levels in stream water during the early phases of storm events may also indicate some desorption of P from resuspended sediments (Svendsen *et al.*, 1995). In the main channel of the Gjern Å *in situ* measurements of stream bed

P indicated that net retention in summer was followed by mobilisation of the retained P in early autumn and further exhaustion of the retained P pool in the following winter (Svendsen *et al.*, 1995).

Analysis of N and P removal in relation to fluxes in lowland streams suggests that stream buffering processes are relatively unimportant in relation to annual budgets in most streams. However, there is considerable evidence to indicate that N and P removal mechanisms are significant in low flow periods particularly during summer months. Removal of nitrate and SRP during base flows has a considerable effect on stream water concentrations of these nutrients and can regulate the timing and form of export to downstream rivers, lakes and estuaries. Although a small proportion of annual N and P flux is transported during stream low flows, the considerable time duration of these conditions enhances the significance of headwater streams as buffers. In temperate streams, low flows often occur on 130-150 days during the warm season of the year. In tropical landscapes denitrification and other N and P sink processes may be important in low flow conditions throughout the year.

MANAGEMENT IMPLICATIONS

Current knowledge of processes such as denitrification and sediment P adsorption that are critical to stream N and P buffering capacity indicate that these processes are strongly influenced by water residence times and the accumulation of fine textured organic rich sediments in streams. Channelisation of small headwater streams to increase drainage capacity tends to reduce water contact time with the stream bed and enhance the removal of fine sediments. In contrast, management manipulations that reduce water velocity such as reductions of channel slope or the construction of small ponds would tend to increase water retention times and the accumulation of organic matter and thus enhance the stream nutrient buffering capacity. The afforestation of stream riparian zones to enhance the buffering capacity of the stream side environment (Petersen *et al.*, 1992; Muscutt *et al.*, 1993) can also improve the efficiency of stream N and P removal by increasing the supply of terrestrial carbon to stream sediments.

Macrophytes play a significant role in nitrate uptake in some streams and can also stimulate denitrification by encouraging significant deposition of organic matter and increasing the supply of nitrate in sediments by nitrification activity in the root zone (Christensen and Sorensen, 1988). The establishment and maintenance of macrophyte communities in small streams unshaded by adjacent riparian vegetation can therefore also increase the stream nutrient removal capacity during low flows.

The costs involved in reduced drainage capability on agricultural land can mitigate against stream management practices that increase water retention times in order to maintain or enhance the nutrient buffering capacity of small headwater streams. It is therefore important to consider the benefits of stream N and P removal processes in making catchment management decisions. It is also essential to integrate the management of stream buffers with other potential buffer zones such as riparian zones and small on-stream ponds as part of a more comprehensive approach to the management of nutrient fluxes within the landscape.

CONCLUSIONS

There is now considerable evidence that streams play a significant role in regulating the flux of N and P during base flows particularly in summer, although the annual capacity to remove N and P is often low. Nevertheless, many aspects of streams as N and P buffers are inadequately understood. The influence of the hyporheic zone on stream water chemistry in relatively eutrophic agricultural streams is still unknown, as is the capacity of this subsurface environment to act as a buffer for nitrate rich groundwater which discharges directly through the stream bed. Further research is needed to evaluate P transport and retention processes in small agricultural streams which do not receive large point sewage inputs. Most studies of nutrient removal in headwater streams are from temperate landscapes and little information is available in tropical environments. A more detailed understanding of the potential role of in-stream and hyporheic environments as buffer zones in a wider range of streams and landscape settings can contribute to the successful management of N and P in catchments.

REFERENCES

Bache, B.W. and Williams, E.G. (1971) A phosphate sorption index for soils. *J. Soil Sci.*, **22**, 289-301.

Bencala, K.E. (1993) A perspective on stream-catchment connections. *J. N. Am. Benthol. Soc.*, **12**, 44-47.

Bohlke, J.K. and Denver, J.M. (1995) Combined use of groundwater dating, chemical, and isotopic analysis to resolve the history and fate of nitrate contamination in two agricultural watersheds, Atlantic coastal plain, Maryland. *Water Res. Res.*, **31**, 2319-2339.

Casey, H. (1977) Origin and variation of nitrate nitrogen in the chalk springs, streams and rivers in Dorset and its utilisation by higher plants. *Prog. Water Technol.*, **8**, 225-235.

Casey, H. and Clarke, R.T. (1986) Monitoring to detect changes in water quality series. *IASH Publ.*, **157**, 257-265.

Chatarpaul, L., Robinson, J.B. and Kaushik, M.K. (1980) Effects of tubificid worms on denitrification and nitrification in stream sediment. *Can. J. Fish. Aquat. Sci.*, **37**, 656-663.

Christensen, P.B. and Sorensen, J. (1986) Temporal variation of denitrification activity in plant-covered littoral sediment from Lake Hampen, Denmark. *Appl. Environ. Microbiol.*, **51**, 1174-1179.

Christensen, P.B. and Sorensen, J. (1988) Denitrification in sediment of lowland streams: regional and seasonal variation in Gelbaek and Rabis Baek, Denmark. *FEMS Microbiol. Ecol.*, **53**, 335-344.

Christensen, P.B., Nielsen, L.P., Revsbech, N.P. and Sorensen, J. (1989) Microzonation of denitrification activity in stream sediments as studied with a combined oxygen and nitrous oxide microsensor. *Appl. Environ. Microbiol.*, **55**, 1234-1241.

Cooke, J.G. and White, R.E. (1987) Spatial distribution of denitrifying activity in a stream draining an agricultural catchment. *Freshwater Biol.*, **18**, 509-519.

Cooke, J.G. and Cooper, A.B. (1988) Sources and sinks of nutrients in a New Zealand hill pasture catchment. III. *Nitrogen. Hydrol. Proc.*, **2**, 135-149.

Cooper, A.B. (1990) Nitrate depletion in the riparian zone and stream channel of a small headwater catchment. *Hydrobiol.*, **202**, 13-26.

Cooper, A.B. and Cooke, J.B. (1984) Nitrate loss and transformation in two vegetated headwater streams. *New Zealand J. Marine and Freshwater Res.*, **18**, 441-450.

Dorioz, J.M., Pillebrue, E. and Ferki, A. (1989) Dynamique du phosphore dans les bassins versants: importance des phenomenes de retention dans les sediments. *Wat. Res.*, **23**, 147-158.

Duff, J.H. and Triska, F.J. (1990) Denitrification in sediments from the hyporheic zone adjacent to a small forested stream. *Can. J. Fish. Aquat. Sci.*, **47**, 1140-1147.

Duff, J.H., Triska, F.J. and Oremland, R.S. (1984) Denitrification associated with stream periphyton: chamber estimates from undisrupted communities. *J. Environ. Qual.*, **3**, 514-518.

Elwood, J.W., Newbold, J.D., Trimble, A.F. and Stark, R.W. (1981) The limiting role of phosphorus in a woodland stream ecosystem: effects of P enrichment on leaf decomposition and primary producers. *Ecology*, **62**, 146-158.

Faafeng, B.A. and Roseth, R. (1993) Retention of nitrogen in small streams artificially polluted with nitrate. *Hydrobiol.*, **251**, 113-122.

Findlay, S. (1995) Importance of surface-subsurface exchange in stream ecosystems: the hyporheic zone. *Limnol. Oceanogr.*, **40**, 159-164.

Fraser, P. and Chilvers, C. (1981) Health aspects of nitrate in drinking water. *Sci. Total Environ.*, **18**, 103-116.

Green, D.B., Logan, T.L. and Smeck, N.E. (1978) Phosphate adsorption-desorption characteristics of suspended sediments in the Maumee river basin of Ohio. *J. Environ. Qual.*, **7**, 208-212.

Grimm, M.B., Fisher, S.G. and Minckley, W.L. (1981) Nitrogen and phosphorus dynamics in hot desert streams of southwestern U.S.A. *Hydrobiol.*, **83**, 303-312.

Harms, L.L., Vidal, P.H. and McDermott, T.E. (1978) Phosphorus interactions with stream bed sediments. *J. of Environ. Engineering Div. ASCE*, **104**, 271-288.

Harvey, J.W. and Bencala, K.E. (1993) The effect of streambed topography on surface-subsurface water exchange in mountain catchments. *Water Resour. Res.*, **29**, 89-98.

Heathwaite, A.L., Johnes, P.J. and Peters, M.E. (1996) Trends in nutrients. *Hydrol. Proc.*, **10**, 263-293.

Hendricks, S.P. and White, D.S. (1995) Seasonal biogeochemical patterns in surface water, subsurface hyporheic, and riparian groundwater in a temperate stream ecosystem. *Arch. Hydrobiol.*, **134**, 459-490.

Henricksen, K., Rasmussen, M.B. and Jensen, A. (1983) Effect of bioturbation on microbial nitrogen transformations in the sediment and fluxes of ammonium and nitrate to the overlaying water. *Ecol. Bull. Stockholm*, **35**, 193-205.

Hill, A.R. (1979) Denitrification in the nitrogen budget of a river ecosystem. *Nature*, **281**, 291-292.

Hill, A.R. (1981) Nitrate-nitrogen flux and utilisation in a stream ecosystem during low summer flows. *Can. Geogr.*, **25**, 226-239.

Hill, A.R. (1982) Phosphorus and major cation mass balances for two rivers during low summer flows. *Freshwater Biol.*, **12**, 293-304.

Hill, A.R. (1983a) Nitrate-nitrogen mass balances for two Ontario rivers. In: Fontaine, T.D. and Bartell, S.M. (Eds.) *Dynamics of Lotic Ecosystems.* p. 457-477. Ann Arbor Science, Ann Arbor, Michigan.

Hill, A.R. (1983b) Denitrification: its importance in a river draining an intensively cropped watershed. *Agric. Ecosyst. Environ.*, **10**, 47-62.

Hill, A.R. (1986) Stream nitrate-N loads in relation to variations in annual and seasonal runoff regimes. *Water Resour. Bull.*, **22**, 829-839.
Hill, A.R. (1988) Factors influencing nitrate depletion in a rural stream. *Hydrobiol.* **160**, 111-122.
Hill, A.R. (1996) Nitrate removal in stream riparian zones. *J. Environ. Qual.*, **25**, 743-755.
Hill, A.R. and Sanmugadas, K. (1985) Denitrification rates in relation to stream sediment characteristics. *Wat. Res.*, **19**, 1579-1586.
Hoare, R.A. (1979) Nitrate removal from streams draining experimental catchments. *Prog. Water Tech.*, **11**, 303-314.
House, W.A. (1990) The prediction of phosphate coprecipitation with calcite in freshwaters. *Wat. Res.*, **24**, 1017-1023.
House, W.A., Denison, F.H. and Armitage, P.D. (1995) Comparison of the uptake of inorganic phosphorus to a suspended and stream bed sediment. *Wat. Res.*, **29**, 767-779.
Howard-Williams, C., Davies, J. and Pickmere, S. (1982) The dynamics of growth, the effects of changing area and nitrate uptake by watercress *Nasturtium officinale* in a New Zealand stream. *J. Appl. Ecol.*, **19**, 589-601.
Howard-Williams, C., Pickmere, S. and Davies, J. (1983) Decay rates and nitrogen dynamics of decomposing watercress (*Nasturtium officinale* R. Br.). *Hydrobiol.*, **99**, 207-214.
Jansson, M., Leonardson, L. and Fejes, J. (1994) Denitrification and nitrogen retention in a farmland stream in southern Sweden. *Ambio*, **23**, 326-331.
Johnson, A.H., Bouldin, D.R., Goyette, E.A. and Hedges, A.M. (1976) Phosphorus loss by stream transport from a rural watershed: quantities, processes and sources. *J. Environ. Qual.*, **5**, 148-157.
Jones, J.B., Fisher, S.G. and Grimm, N.B. (1995) Nitrification in the hyporheic zone of a desert stream ecosystem. *J. N. Am. Benthol. Soc.*, **14,** 249-258.
Kaushik, M.K. and Robinson, J.B. (1976) Preliminary observations on nitrogen transport during summer in a small spring-fed Ontario stream. *Hydrobiol.*, **49**, 59-63.
Kaushik, M.K., Robinson, J.B., Sain, P., Whiteley, H.R. and Stammers, W. (1975) A quantitative study of nitrogen loss from water of a small, spring-fed stream. *Water Pollut. Res. Canada*, **10**, 110-117.
Klotz, R.L. (1985) Factors controlling phosphorus limitation in stream sediments. *Limnol. Oceanogr.*, **30**, 543-553.
Klotz, R.L. (1988) Sediment control of soluble reactive phosphorus in Hoxie Gorge Creek, New York. Can. *J. Fish. Aquat. Sci.*, **45**, 2026-2033.
Klotz, R.L. (1991) Temporal relation between soluble reactive phosphorus and factors in stream water and sediments in Hoxie Gorge Creek, New York. *Can. J. Fish. Aquat. Sci.*, **48**, 84-89.
McCallister, D.L. and Logan, T.J. (1978) Phosphate adsorption-desorption characteristics of soils and bottom sediments in the Maumee River basin of Ohio. *J. Env. Qual.*, **7**, 87-92.
McColl, R.H.S. (1974) Self-purification of small freshwater streams: phosphate, nitrate and ammonia removal. N. Z. *Jour. of Marine and Freshw. Res.*, **8**, 375-388.
Meyer, J.L. (1979) The role of sediments and bryophytes in phosphorus dynamics in a headwater stream ecosystem. *Limnol. Oceanogr.*, **24**, 365-375.
Mulholland, P.J., Newbold, J.D., Elwood, J.W., Ferren, L.A. and Webster, J.R. (1985) Phosphorus spiralling in woodland streams: seasonal variations. *Ecology*, **66**, 1012-1023.
Munn, M.L. and Meyer, J.L. (1990) Habitat-specific solute retention in two small streams: an intersite comparison. *Ecology*, **71**, 2069-2082.
Newbold, J.D. (1994) Cycles and spirals of nutrients. In: Calow, P. and Petts, G.E. (Eds.) *The Rivers Handbook.* p. 379-395. Blackwell Scientific Publ. Oxford.
Newbold, J.D., Elwood, J.W., O'Neill, R.V. and VanWinkle, W. (1981) Measuring nutrient spiralling in streams. *Can. J. Fish. Aquat. Sci.*, **38**, 860-863.
Muscutt, A.D., Harris, G.L., Bailey, S.W. and Davies, D.B. (1993) Buffer zones to improve water quality: a review of their potential use in UK agriculture. *Agric. Ecosyst. Environ.*, **45**, 59-77.
Petersen, R.C., Petersen, L. B-M. and Lacoursiére, J. (1992) In: Boon, P.J., Calow, P. and Petts, G.E. (Eds.) *River Conservation and Management.* p. 293-309. John Wiley and Sons Ltd.
Pinay, G., Haycock, N.E., Ruffinoni, C. and Holmes, R.M. (1994) The role of denitrification in nitrogen removal in river corridors. In: Mitsch, W.J. (Ed.) *Global Wetlands: Old World and New.* p. 107-116. Elsevier Science B.V.
Phillips, P.J., Denver, J.M., Shedlock, R.J. and Hamilton, P.A. (1993) Effect of forested wetlands on nitrate concentrations in groundwater and surface water on the Delmarva peninsula. *Wetlands*, **13**, 75-83.
Robertson, W.D., Cherry, J.A. and Sudicky, E.A. (1991) Ground water contamination from two small septic systems on sand aquifers. *Ground Water*, **29**, 82-92.
Robinson, J.B., Whiteley, H.R., Stammers, W., Kaushik, N.K. and Sain, P. (1979) The fate of nitrate in small streams and its management implications. In: Loehr, R.C., Haith, D.H., Walter, M.F. and Martin, C.S. (Eds.) *Best Management Practices for Agriculture and Silviculture.* p. 247-259. Ann Arbor Science. Ann Arbor Michigan.
Sain, P., Robinson, J.B., Stammers, W.M., Kaushik, N.K. and Whiteley, H.R. (1977) A laboratory study of the role of stream sediment in nitrogen loss from water. *J. Env. Qual.*, **6**, 276-278.
Sebetich, M.J., Kennedy, V.C., Zand, S.M., Avanzino, R.J. and Zellweger, G.W. (1984) Dynamics of added nitrate and phosphate compared in a northern California woodland stream. *Water Res. Bull.*, **20**, 93-101.

Shulka, S.S., Syers, J.K., Williams, J.D.H., Armstrong, D.E. and Harris, R.F. (1971) Sorption of inorganic phosphate by lake sediments. *Soil Sci. Soc. Amer. Proc.*, **35**, 244-249.
Stainton, M.P. (1980) Errors in molybdenum blue methods for determining orthophosphate in freshwater. *Can. J. Fish. Aquat. Sci.*, **37**, 472-478.
Stanford, J.A. and Ward, J.V. (1988) The hyporheic habitat of riverine ecosystems. *Nature,* **335**, 64-66.
Stone, M. and Mudroch, A. (1989) The effect of particle size, chemistry and mineralogy of river sediments on phosphate adsorption. *Env. Tech. Lettr.*, **10**, 501-510.
Stone, M., English, M.C. and Mulamootil, G. (1991) Sediment and nutrient transport dynamics in two tributaries of Lake Erie: a numerical model. *Hydrol. Proc.*, **5**, 371-382.
Stone, M., Mulamootil, G. and Logan, L. (1995) Grain size distribution effects on phosphate sorption by fluvial sediment: implications for modelling sediment-phosphate transport. *Hydrol. Sci. J.*, **40**, 67-81.
Svendsen, L.M. and Kronvang, B. (1993) Retention of nitrogen and phosphorus in a Danish lowland river system: implications for the export from the watershed. *Hydrobiol.*, **251**, 123-135.
Svendsen, L.M., Kronvang, B., Kristensen, P. and Graesbol, P. (1995) Dynamics of phosphorus compounds in a lowland river system: importance of retention and non-point sources. *Hydrol. Proc.*, **9**, 119-142.
Taylor, A.W. and Kunishi, H.M. (1971) Phosphate equilibria on stream sediment and soil in a watershed draining an agricultural region. *J. Agr. Food. Chem.* **19**, 827-831.
Tiedje, J.M. (1982) Denitrification. In: Page, A.L. (Ed.) *Methods of Soil Analysis Part 2. Chemical and microbiological properties.* p. 1011-1026. Agronomy Monograph No. 9, 2nd ed. ASA-SSSA, Madison, WI.
Thibodeaux, L.J. and Boyle, J.D. (1987) Bedform generated convective transport in bottom sediment. *Nature,* **325**, 341-343.
Triska, F.J. and Oremland, R.S. (1981) Denitrification associated with periphyton communities. *Appl. Environ. Microbiol.*, **42**, 745-748
Triska, F.J., Kennedy, V.C., Avanzino, R.J., Zellweger, G.W. and Bencala, K.E. (1989a) Retention and transport of nutrients in a third-order stream in northwestern California: hyporheic processes. *Ecology,* **70**, 1893-1905.
Triska, F.J., Kennedy, V.C., Avanzino, R.J., Zellweger, G.W. and Bencala, K.E. (1989b) Retention and transport of nutrients in a third order stream: channel processes. *Ecology,* **70**, 1877-1892.
Triska, F.J., Duff, J.H. and Avanzino, R.J. (1993) Patterns of hydrological exchange and nutrient transformation in the hyporheic zone of a gravel-bottom stream: examining terrestrial-aquatic linkages. *Freshwater Biol.*, **29**, 259-274.
Valett, H.M., Fisher, S.G. and Stanley, E.H. (1990) Physical and chemical characteristics of the hyporheic zone of a Sonoren desert stream. *J.N. Am. Benthol. Soc.*, **9**, 201-215.
Valett, H.M., Hakenkamp, C.C. and Boulton, A.J. (1993) Perspectives on the hyporheic zone: integrating hydrology and biology: Introduction. *J.N. Am. Benthol. Soc.*, **12**, 40-43.
Valett, H.M, Fisher, S.G., Grimm, N.B. and Camill, P. (1994) Vertical hydrologic exchange and ecological stability of a desert stream ecosystem. *Ecology,* **75**, 548-560.
Valett, H.M., Morrice, J.A., Dahm, C.N. and Campana, M.E. (1996) Parent lithology, surface-groundwater exchange and nitrate retention in headwater streams. *Limnol. Oceanogr.*, **41**, 333-345.
Ventullo, R.M. and Rowe, J.J. (1982) Denitrification potential of epilithic communities in a lotic environment. *Current Microbiol.*, **7**, 29-34.
Vincent, W.F. and Downes, M.T. (1980) Variations in nutrient removal from a stream by watercress *(Nasturtium officinale). Aquat. Bot.*, 9, 221-235.
Walling, D.E. (1971) Streamflow from instrumental catchments in south-east Devon. In: Gregory, K.J. and Ravenhill, W.L.D. (Eds.) *Exeter Essays in Geography.* p. 55-81. University of Exeter Press, England.
White, D.S. (1993) Perspectives on defining and delineating hyporheic zones. *J. N. Am. Benthol. Soc.*, **12**, 61-69.
White, D.S., Hendricks, S.P. and Fortner, S.L. (1992) Groundwater-surface water interactions and the distributions of aquatic macrophytes. In: Standford, J.A. and Simons, J.J. (Eds.) *Proceedings of the First International Conference on Ground Water Ecology.* p. 267-256 American Water Resources Association, Bethesda, Maryland.
Wyer, M.D. and Hill, A.R. (1984) Nitrate transformations in southern Ontario stream sediments. *Water Resour. Bull.*, **20**, 729-737.

13 The Potential Role of Riparian Forests as Buffers Zones

R. Lowrance

USDA-Agricultural Research Service (ARS), Tifton, GA 31793, USA.

Abstract

Riparian forest buffer systems provide effective control of non-point source pollution in some settings. Pollution control is dependent on the type of pollutant, the hydrologic connection between pollution sources and the riparian buffer, and the characteristics of the riparian soils and vegetation. Knowledge of these complex interacting factors is generally incomplete and use of riparian forest buffers requires professional judgement regarding where to expect effective non-point source pollution control. These professional judgements should be based on best available information on flow systems and the behaviour of natural, managed and restored riparian ecosystems in the region of interest. Factors related to establishment, sustainability, management, maintenance, loading rates and size of the buffer system must also be considered. Results of a Best Professional Judgement process to assess the potential pollution control by riparian forest buffers in the Chesapeake Bay Watershed concluded that pollution control would range from very high levels in some coastal plain systems to very low levels in systems dominated by regional groundwater.

INTRODUCTION

Application of research to broad-scale environmental issues often requires using professional judgement when faced with inadequate knowledge or incomplete understanding. Public policies to encourage or require landscape management techniques such as riparian (streamside) management will often need to proceed with Best Professional Judgement (BPJ) decisions based on both site-specific experiments and fundamental understanding of hydrology, soils and ecosystem function.

Riparian forest buffer systems (RFBS) are streamside ecosystems, managed for the enhancement of water quality through control of non-point source pollution (NPS) and protection of the stream environment. The use of riparian management zones is relatively well-established as a Best Management Practice (BMP) for water quality improvement in forestry practices (Comerford *et al.*, 1992), but has been much less widely applied as a BMP in agricultural areas or in urban or suburban settings. Riparian ecosystems are especially important on small streams (1st, 2nd and 3rd order) which account for over three-quarters of the total stream length in the United States (Leopold *et al.*, 1964). Riparian forests of mature trees (30 to 75 yrs. old) are known to effectively reduce non-point source pollution from agricultural fields in certain landscapes (Hanson *et al.*, 1994; Lowrance *et al.*, 1984; Peterjohn and Correll, 1984; Jacobs and Gilliam, 1985). RFBS specifications have been proposed by the United States Department of Agriculture-Forest Service (USDA-FS) and the USDA-Natural Resources Conservation Service (Welsch, 1991; NRCS, 1995): Zone 1 of the RFBS is permanent woody vegetation immediately adjacent to the stream bank; Zone 2 is managed forest occupying a strip upslope from Zone 1; Zone 3 is an herbaceous filter strip upslope from Zone 2. The specification applies to areas where cropland, grasslands and/or pasture are adjacent to riparian areas on permanent or intermittent streams, margins of lakes and ponds, margins of wetlands or margins of groundwater recharge areas such as sinkholes.

The three-zone RFBS specification is based on studies of naturally occurring riparian forests along low order (1st to 4th order) streams and experimental scale grass filter strips. Under natural conditions, riparian forest ecosystems formed a dynamic yet stable buffering system along most shorelines, rivers and streams in many regions, worldwide. Although few studies have documented the specific changes in water quality functions during the establishment period of a riparian forest, established

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0

RFBS are expected to sustain water quality functions over the long term in a manner similar to natural riparian systems.

The functions of riparian systems to control NPS pollution are dependent on hydrologic connection(s) of pollutant source(s) with the RFBS. Although generalisations can be made, the extent, timing and spatial variability of the hydrologic connections add uncertainty to assessment of NPS pollution control. It is critical to understand what proportion of the pollutant moves through the biologically active soil and litter layers. The hydrologic connection between source areas and riparian ecosystems ranges from nearly 100% of the water moving across the surface or in shallow groundwater to a very low percentage of flow moving through riparian ecosystems. This lower limit is not well-defined, but a conservative estimate can be made by hydrograph analysis to separate storm flow from baseflow. At a minimum, most stormflow should move in either surface runoff or shallow groundwater and should be subject to processing in a RFBS. Additionally, the difference in maximum baseflow and minimum baseflow is another useful indicator of the proportion of water available for riparian processing.

MANAGEMENT CONSIDERATIONS

Loading Rates

Higher rates of nitrate removal should be possible under higher loadings of nitrate especially where denitrification (microbial reduction of nitrate to nitrogen gases) is the primary means of nitrate removal. Given the range in nutrient uptake possible both among different plant species and within the same plant species, it is likely that vegetation uptake will increase with increasing loads if there is significant hydrologic interaction with vegetation.

Increasing loads of P are likely to be less effectively controlled than increasing loads of N, because of the lack of a microbial process analogous to denitrification to remove or sequester P in the RFBS. If increasing P loads are to be controlled, it will require effective management of Zones 3 and 2 for sediment removal and infiltration. If dissolved or particulate P can be retained in the root zone, it will be available for both biological and chemical removal processes. If RFBS have some absolute removal potential for P, reducing input loads should increase the efficiency of removal. The ability of Zone 2 to retain P may be limited, especially under high loadings of dissolved P.

Management to control increasing loads of sediment and sediment-borne chemicals will require specific management of Zones 3 and 2 for sediment retention. Most of the mass of sediment will be deposited in Zone 3 and most of the sediment-borne nutrients will be deposited in Zone 2. Increased sediment loadings to Zone 3 will require increased management to eliminate concentrated flows, remove accumulated sediment (especially in berms) and restore the herbaceous vegetation. Increased sediment and sediment-borne chemicals to Zone 2 should lead to higher amounts of chemical deposition in surface litter.

Loading rate/buffer size relationships are only poorly defined, especially for dissolved pollutants. With water in contact with surface litter or the biologically active root zone, buffers of about 30 m have been effective for at least sediment and nitrate removal (Hanson *et al.*, 1994; Lowrance *et al.*, 1984; Peterjohn and Correll, 1984; Jacobs and Gilliam, 1985). Franklin *et al.* (1996) found that a field area: forest buffer area ratio of 5.8 was an effective buffer but a ratio of 11.1 was not effective. One of the difficulties in describing these relationships is that increasing pollutant loads may also be accompanied by increasing water volumes in either surface runoff, groundwater, or both. In the presence of increased water movement, denitrification for nitrate removal should be enhanced and sedimentation and infiltration may be decreased. Increased surface runoff and loading of sediment and sediment-borne chemicals can be accommodated by management of Zones 3 and 2 to increase roughness and control channelised flow. Although mass balance approaches may be extrapolated to higher loading rates, they provide only an estimate and may not predict real-world responses.

Stream Order and Stream Size

Regardless of the size of stream or the hydrologic setting, water moving across the surface or through the root zone of a RFBS should show reduction in either nitrate (groundwater) or sediment and sediment-borne chemical loads reaching the stream (surface runoff). As streams increase in size, the integrated effects of immediately adjacent riparian ecosystems decreases relative to the overall water quality of the stream. On lower order streams there is greatest potential for interactions between water and riparian areas. For NPS pollution control, the change in impact of RFBS as stream order increases can be estimated based on hydrologic contributions from upstream and from the riparian ecosystem. For first-order streams, the potential impact of the RFBS on chemical load or flow-weighted concentration is directly proportional to the proportion of the excess precipitation from the contributing area which moves through or near the root zone or surface of the RFBS. For all streams above first order, the contributing area is only one source of pollutants, with upstream reaches providing the other source. For second-order and above, the NPS pollution control function of a given RFBS is based on both the proportion of water from the contributing area which moves through the riparian system and the relative sizes of the two potential pollutant loads: upstream sources or adjacent land uses. Clearly, the larger the stream, the less impact a RFBS along a particular stream reach can have on reduction in overall load within that reach.

On a watershed basis, the higher the proportion of total streamflow originating from relatively short flow-paths to small streams, the larger the potential impact of RFBS. In comparing the potential effectiveness of RFBS among watersheds, drainage density (length of channel per unit area of watershed) should provide a useful starting point. Higher drainage density implies greater potential importance for RFBS in NPS pollution control.

Establishment and Sustainability

RFBS should be used as part of an integrated land management or conservation system which consists of 1) watershed scale management, 2) NPS pollution management, and 3) active management of the RFBS. In this way, RFBS become part of conservation, stormwater, nutrient and farm management, timber harvest and other land management planning efforts.

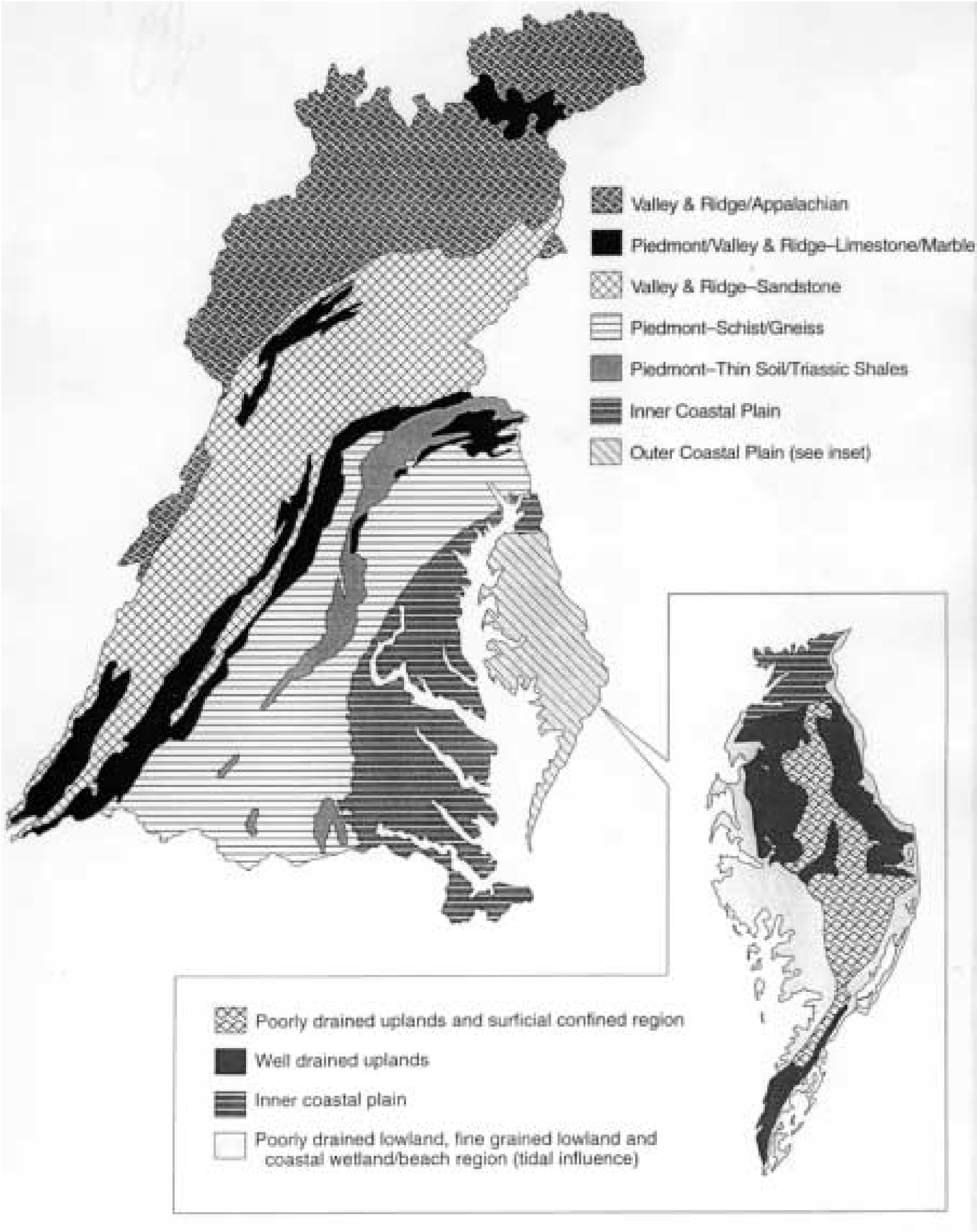
Watershed management is essential to reduce overall pollutant loadings and integrate the riparian area as part of a landscape influenced by upstream hydrology. In a landscape context, RFBS which mimic the natural ecosystems of the area will increase the likelihood of long-term sustainability. Consideration of existing riparian forests and linkage of RFBS as continuous stream corridors is desirable. Source management and land conservation measures are important in conserving natural resources, reducing overall pollution and limiting stress on the RFBS. These measures, along with maintenance of buffer plantings, are especially important during the establishment phase and in preventing excessive runoff or sediment and nutrient loading beyond the capacity of the buffer. RFBS management such as periodic harvesting, runoff control maintenance, control of invasive plants, etc. is desirable to maximise performance and ensure long-term effectiveness. Continued runoff control and protection of Zone 1 functions are essential to maintaining optimum performance in RFBS.

Integration of RFBS within land management helps to prevent some of the primary reasons for "acute" failure such as runoff inputs which exceed the design of the RFBS and cut gullies or channels, or failure to address "chronic" problems such as a gradual decrease in phosphorus retention. Because of the commitment of land required for RFBS establishment, the approaches used for establishment and subsequent management should contribute to RFBS which are sustainable for decades.

CASE STUDY – CONTRASTING CONDITIONS IN A WATERSHED

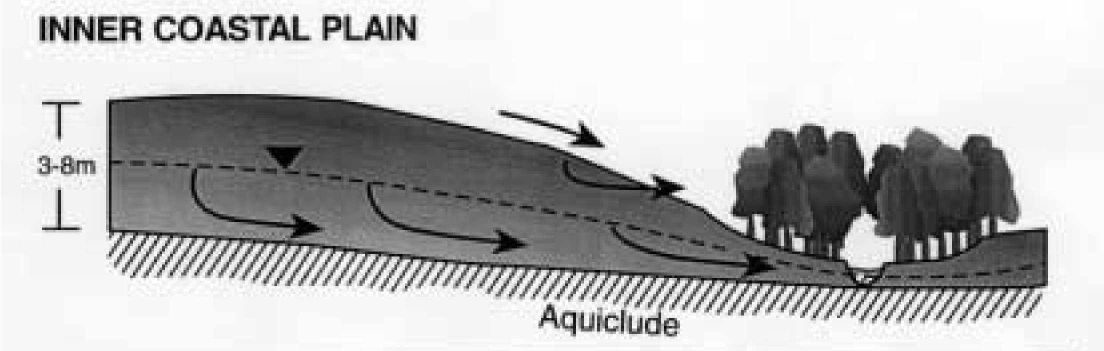
Portions of the Chesapeake Bay Watershed (Figure 1) (US) have been evaluated for potential NPS pollution control by RFBS (Lowrance *et al.*, 1995). For example, two portions of the Gulf-Atlantic Coastal Plain were judged to have very different potential for removing nitrate from groundwater before reaching streams. The Inner Coastal Plain (Figure 2) is expected to have nearly 100% nitrate

Figure 1. Physiographic regions of the Chesapeake Bay Watershed with inset showing details of the outer Coastal Plain.



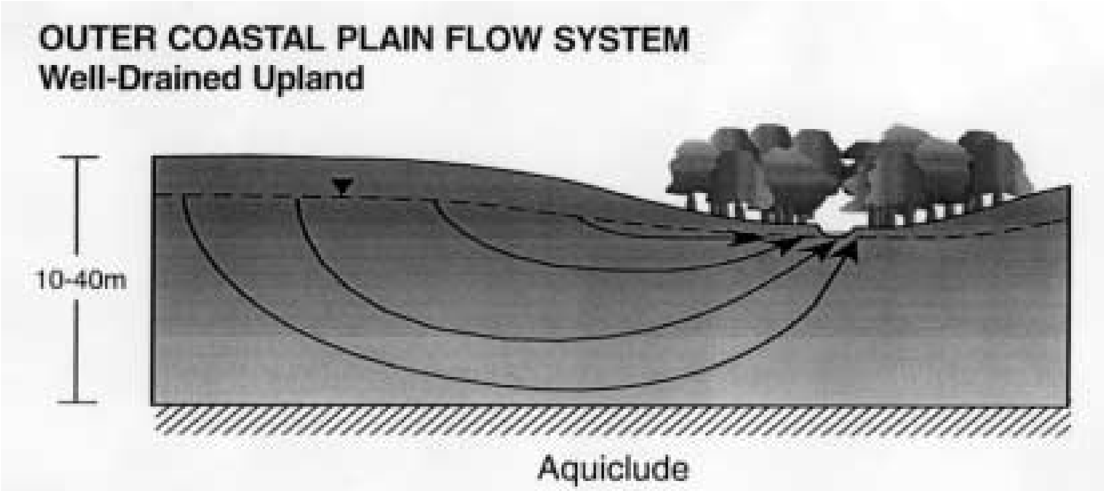
removal because most flow takes place along relatively short flow paths which pass through or near the root zone. In contrast, a portion of the Outer Coastal Plain designated the Well-Drained Upland (Figure 3) was judged to have very little potential for removal of nitrate due to long flow paths which move water below the riparian ecosystem root zone and discharge directly to the stream channel.

Figure 2. Generalised flow system of the Inner Coastal Plain.



Water Quality Function	Expected Level	Critical Constraints	Restoration/ Enhancement
Removal of nitrate from groundwater	High, most water moves in or near root zone.	Bypass due to artificial subsurface drains. Organics in Zone 2.	Important on all streams. Rapid restoration of dentrification function. Ground cover in Zone 3.
Removal of sediment and sediment-borne pollutants	High/Medium	Convert concentrated flow to sheet flow.	Restore in all areas. Enhance existing forest with Zone 3 speaders.
Removal of dissolved phosphorus	Medium/Low	Control of dissolved P in surface runoff and groundwater is limited.	Restore in areas with major P load in surface runoff. Enhance existing forest with Zone 3.

Figure 3. Generalised flow system of the Outer Coastal Plain; Well-Drained Upland.



Water Quality Function	Expected Level	Critical Constraints	Restoration/ Enhancement
Removal of nitrate from groundwater	Low, primarily removal from shorter flow paths.	Bypass flow due to deeper aquifers. Long flow paths surface in stream channels.	Concentration on headwater areas. Zone 1 important for nitrate removal.
Removal of sediment and sediment-borne pollutants	High/Medium	Concentrated flow must be converted to sheet flow.	On larger streams, focus on filtering eroded sediment. Enhance functions of Zones 2 & 3.
Removal of dissolved phosphorus	Medium/Low	Dissolved P control is limited. Focus on P load in surface runoff.	Increase vegetation uptake and accretion. Enhance existing forest and grass strips.

For the Inner Coastal Plain, BPJ on nitrate control predicts:

1) Based on mass balances, established RFBS remove 20 to 40 kg NO_3-N ha^{-1} yr^{-1} from subsurface flow. For the systems studied, this was 85-90% retention of nitrate.

2) For the RFBS to be applicable in systems with artificial drainage near streams, the drainage system will have to be modified to work in conjunction with the RFBS.
3) Newly-established systems are likely to have a substantial effect on subsurface nitrate loads in (at most) 5 to 10 years if anoxic sediments and high organic matter surface soils are already in place. By 15 to 20 years, re-established RFBS should control groundwater nitrate loads in most situations.
4) The nitrate concentration data indicated that higher nitrate loadings could be removed in the RFBS if it was exposed to higher loadings than represented in the mass balance studies. This is most likely to be true in systems with highest denitrification rates or potentials.

For the Outer Coastal Plain, BPJ on nitrate control predicts:

1) In most areas, there will not be strong hydrologic connections between nitrate sources and riparian forests, resulting in little removal of nitrate. Where hydrologic connections between groundwater and biologically active soil layers are made, RFBS in the Outer Coastal Plain should have about the same capacity for nitrate removal as in the Inner Coastal Plain.
2) The Zone 1 vegetation (adjacent to the stream channel) should be very important because of potential access to water and pollutants in the hyporheic zone. Zone 1 vegetation should be managed for N uptake and for formation of high organic matter surface soils. Provision of leaf litter and other organic matter to the stream channels may increase denitrification in the channel and hyporheic zone.
3) Re-establishment of RFBS in the Outer Coastal Plain should focus on headwater streams, many of which have been ditched. Enhancement of existing forests along both small and large streams should focus on management of Zone 1 to intercept nitrate enriched groundwater.

SUMMARY AND CONCLUSIONS

Predicting the usefulness of riparian forest buffers for pollution control will need to be done with a Best Professional Judgement approach until more detailed studies are available from a wide range of watersheds and land uses. This approach depends on knowledge of site conditions, loading rates of pollutants and ecosystems functions. Best Professional Judgements based on potential or actual hydrologic connections between pollutant source and the riparian forest buffer provide a means for regional scale assessment of the pollution control by riparian forest buffers.

REFERENCES

Comerford, N.B., Neary, D.G. and Mansell, R.S. (1992) *The effectiveness of buffer strips for ameliorating offsite transport of sediment, nutrients and pesticides from silvicultural operations.* Natl. Coun. Paper Ind. Air & Stream Improvement Tech. Bull. 631. New York, NY.

Franklin, E.C., Gregory, J.D., Parsons, J.E. and Hazel, D.W. (1996) *Management of forested filter zones for dispersion and treatment of agricultural runoff.* Univ. N. Carolina Water Resour. Res. Inst., Raleigh, NC.

Hanson, G.C., Groffman, P.M. and Gold, A.J. (1994) Denitrification in riparian wetlands receiving high and low groundwater nitrate inputs. *J. Environ. Qual.,* **23**, 917-922.

Jacobs, T.C. and Gilliam, J.W. (1985) Riparian losses of nitrate from agricultural drainage waters. *J. Environ. Qual.,* **14**, 472-478.

Leopold, L.B., Wolman, M.G. and Miller, J.P. (1964) *Fluvial processes in geomorphology.* W.H. Freeman and Company, San Francisco.

Lowrance, R.R., Todd, R.L., Fail Jr., J., Hendrickson Jr., O., Leonard, R. and Asmussen, L. (1984) Riparian forests as nutrient filters in agricultural watersheds. *Bioscience,* **34**, 374-377.

Lowrance, R., Altier, L.S., Newbold, J.D., Schnabel, R.R., Groffman, P.M., Denver, J.M., Correll, D.L., Gilliam, J.W., Robinson, J.L., Brinsfield, R.B., Staver, K.W., Lucas, W. and Todd, A.H. (1995) *Water quality functions of riparian forest buffer systems in the Chesapeake Bay Watershed.* Ches. Bay Prog. Rep. 134/95, EPA 903-R-95-004.

NRCS (1995) *Riparian Forest Buffers, 392.* NRCS Watershed Science Institute. Seattle, WA.

Peterjohn, W.T. and Correll, D.L. (1984) Nutrient dynamics in an agricultural watershed: Observation on the role of a riparian forest. *Ecology,* **65**, 1466-1475.

Welsch, D.J. (1991) *Riparian Forest Buffers.* USDA-FS Pub. No. NA-PR-07-91. USDA-FS, Radnor, PA.

14

Effectiveness of buffer strips for attenuation of ammonium and nitrate levels in runoff from pasture amended with cattle slurry or inorganic fertiliser

A. Núñez Delgado, E. López Períago and F. Díaz-Fierros

Department Edafoloxía e Química Agrícola, Fac. Farmacia, Campus Univ. S/N, 15706 Santiago de Compostela, Spain

Abstract

Two adjacent plots marked out on a pasture with 15% mean slope received cattle slurry or inorganic fertiliser at doses selected such that the total amount of nitrogen applied per unit area was the same for each plot. Within each plot, two subplots were marked out: slurry/fertiliser was applied to an area with 8 m downslope length in one subplot, and to an area with 3 m downslope length in the other. At 1, 7 and 21 days after amendment, the plots received simulated rainfall. Runoff was collected in Gerlach troughs situated at various distances downslope (i.e. after passage across buffer strips of different downslope lengths), and analysed for ammonium and nitrate content. In general, runoff percentages were higher downslope of the slurry-amended subplots than downslope of the fertiliser-amended subplots. Likewise, concentrations of the two contaminants (particularly ammonium) were higher in runoff from the slurry-amended subplots. Absolute masses of ammonium and nitrate in runoff from the slurry-amended subplots were closely related to runoff percentages; in addition, absolute masses in runoff after the second and third rainfall events indicated downslope movement of contaminants already removed from the amended areas. Absolute masses in runoff from the fertiliser-amended plots provided evidence of attenuation of contaminant levels by the buffer strip areas. Considering data for runoff from the slurry-amended subplots, strong correlations were observed between runoff percentages and absolute masses after each of the simulated rainfall events; when data for the fertiliser-amended subplots were considered, strong correlations were observed except after the third rainfall event. Percentage attenuation of contaminant levels in runoff from the slurry-amended subplots (with respect to contaminant levels in the slurry) was greater than 90% in almost all cases.

INTRODUCTION

The use of vegetated buffer strips for the protection of watercourses from diffuse-source pollution has increased in recent years (see Nuñez *et al.*, 1995). Despite this, relatively little detailed information is available as regards the efficiency of buffer strips for removing contaminants present in runoff. Coyne *et al.* (1995) have investigated the efficiency of removal of bacteria, while in our region (Galicia, Northwest Spain) there have been studies of the efficiency of removal of pesticides (Basanta *et al.*, 1995) and of contaminants present in cattle slurry (Carballas *et al.*, 1990; Díaz-Fierros *et al.*, 1990; Núñez *et al.*, 1991). Studies of this type need to take into account the biodegradation of adsorbed substances (see Scow *et al.*, 1995) and between-substance interactions (see Castilho *et al.*, 1993).

In Galicia, cattle slurry is a particularly important cause of diffuse pollution (Carballas *et al.*, 1990; Díaz-Fierros *et al.*, 1993). In the present study we compared nitrate and ammonium levels in runoff from plots amended with cattle slurry or a nitrogen (N) fertiliser and evaluated the efficiency of grass buffer strips for attenuating the levels of these contaminants.

Sufficient information now exists to design and restore riparian areas to fully utilise their capabilities in new resource management initiatives to protect water resources. This paper broadly examines

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0

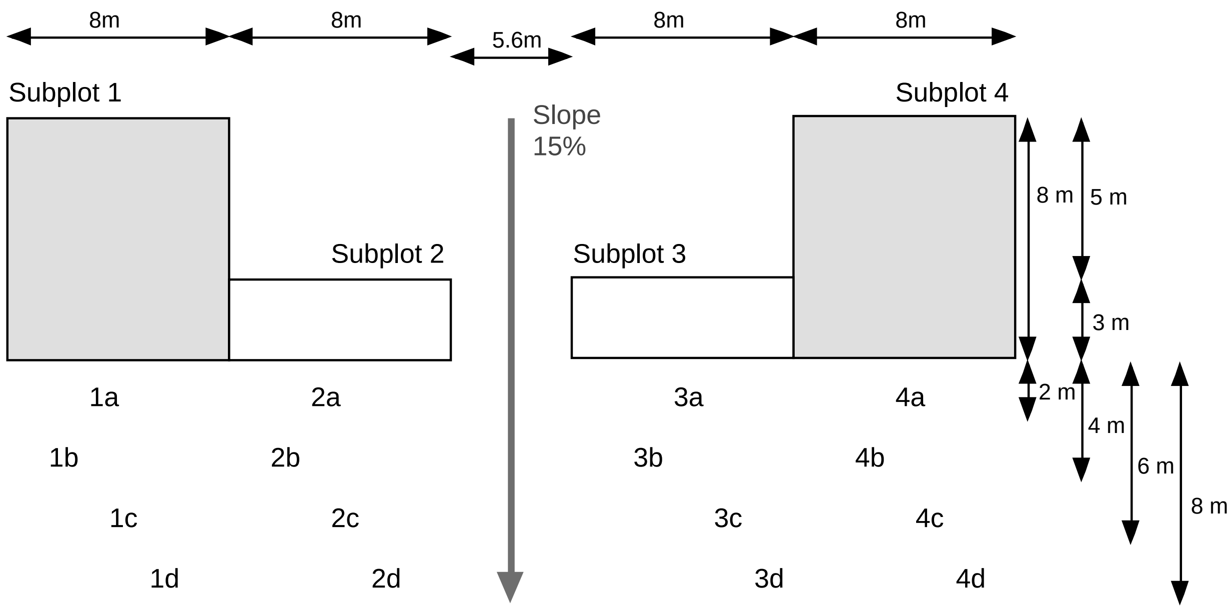
the economic, social and ecological aspects of two riparian forest ecorestoration policy options (25 m v 75 m widths) developed to reduce subsurface nitrate movement into surface waters in Illinois (USA), and demonstrates that agroenvironmental policy can and should be based on ecosystem-level management strategies linked to specific ecosystem functional criteria.

MATERIALS AND METHODS

Study plots

The study was carried out on a site of 15% mean slope, near the village of Sergude (Boqueixón, La Coruña Province, Spain). This site was traditionally cropped but has recently been given over to *Lolium perenne* pasture. Basic characteristics of the A horizon of the soil at this site are: $pH_{(water)}$= 6.1, Carbon(%)= 7.96, Density = 2.5123 g cm^{-3}, Texture = silty loam, cation exchange capacity (CEC)= 11.627 $cmol_c$ kg^{-1} (see more details in Basanta *et al.*, 1995). Two plots destined for slurry or fertiliser application were marked out (Fig. 1). Within each plot, slurry or inorganic fertiliser was applied to subplots of 3 m or 8 m downslope length (Fig. 1). Surface runoff was collected in Gerlach troughs located at various distances downslope of the amended plots (Fig. 1).

Figure 1. Layout of the study plot. Subplots 1 and 2 received cattle slurry (341 m^3 ha^{-1}) and subplots 3 and 4 received inorganic fertilizer (881 kg ha^{-1}). Runoff collection troughs (1a-4d) were located 2 m (a), 4 m (b), 6 m (c) or 8 m (d) downslope of the bottom of the amended plots.



Slurry / fertiliser application

The inorganic fertiliser used was Nitramon (Spain) – i.e. ammonium nitrate NH_4NO_3. Cattle slurry was obtained from a local supplier (pumped from a settling basin); the basic physicochemical characteristics of the batch used (determined as per APHA, 1989) were: pH= 7.06, NH_4+= 701 mg L^{-1}, NO_3-= 32 mg L^{-1}, COD=17700 mg O_2 L^{-1}. The amounts of fertiliser and slurry applied were adjusted so that all amended areas received the same amount of total nitrogen (188.5 kg ha^{-1}). The total N content of Nitramon was determined as 21.4% w/w; fertiliser was thus applied at 880.7 kg ha^{-1}. The total N content of the slurry batch used was determined as 552.64 mg l^{-1} (i.e. g m^{-3}); slurry was thus applied at 340.91 m^3 ha^{-1}. Analyses indicated that the nitrogen in Nitramon is 50% ammonium N and 50% nitrate N, whereas the nitrogen in the slurry was 98.7% ammonium N and only 1.3% nitrate N.

Slurry was sprayed onto plots and dose was controlled by adjusting the duration of the application. Fertiliser was scattered onto plots by hand and samples were taken to check the application rate. In both cases, care was taken to ensure that application was homogeneous.

Rainfall simulation
Rainfall was simulated using two identical spray units in conjunction, as previously described by Basanta *et al.* (1995). Each unit comprises a support structure with four Fulljet 172 HHSS40WSQ wide-angle square-jet nozzles each held at 2 m from the centre pole. Nozzle height is adjustable up to 5.5 m; in this experiment they were set at 4.5 m. Water was pumped from a nearby non-polluted source and its pressure controlled by a manometer for each unit. A pressure of 0.75 kg cm^{-2} (giving an average rainfall intensity of 47 mm h^{-1}) was used. A more complete description of this simulator, and of the characteristics of the simulated rainfall and resulting runoff, is given in Nuñez (1993).

Simulated rainfall was applied 1, 7 and 21 days after slurry/fertiliser application. Each simulated rainfall event consisted of four stages: the two units were first set up on the left of the slurry-amended plot, then moved to the right of this plot; this process was then repeated with the fertiliser-amended plot.

Data collection
The total amount of runoff collected in each trough during and after each simulated rainfall event was determined and runoff samples were analysed for ammonium and nitrate as per Bremner & Keeney (1965).

RESULTS

Runoff percentages
Runoff percentages (between 0.01% and 28.51%) were generally higher downslope of the slurry-amended subplots than downslope of the fertiliser-amended subplots. These differences were particularly marked for the first simulated rainfall event (1 day after slurry/fertiliser application), which suggests that the slurry temporarily blocked pores at the soil surface, leading to reduced infiltration rate.

There was no consistent pattern of differences between runoff percentages downslope of the 8-metre-long subplots and downslope of the 3-metre-long subplots, regardless of whether the slurry- or fertiliser-amended subplots are considered. Runoff percentages were very high for one of the collector troughs (trough 1d), since this trough was situated in a pronounced gully. As pointed out by Dickey & Vanderholm (1981) and Dillaha *et al.* (1986), localised channel flow is likely to reduce the effectiveness of buffer strips for contaminant removal.

Ammonium and nitrate contents in runoff
As expected, ammonium concentrations in runoff collected downslope of the slurry-amended subplots after the first rainfall event were much higher (between 0.19 and 20.62 mg l^{-1}) than in runoff collected downslope of the fertiliser-amended subplots after that event (between 0 and 0.19 mg l^{-1}). Again as expected, nitrate concentrations were higher in runoff collected downslope of the fertiliser-amended subplots (1.60 to 2.88 mg l^{-1}) than in the slurry area (0.32 to 1.60 mg l^{-1}). After the second and third rainfall events, the differences between the slurry- and fertiliser-amended plots were much less pronounced. Indeed, in some cases nitrate level was higher in runoff collected downslope of the slurry-amended subplots than in the corresponding sample collected downslope of the fertiliser-amended subplots; this is probably attributable to oxidation of slurry-derived ammonium to nitrates (as is supported by the observation that ammonium levels in runoff collected downslope of the slurry-amended subplots tended to decline over time). By contrast, ammonium levels in runoff collected

downstream of the fertiliser-amended subplots showed a tendency to increase over time; this is probably because the fertiliser is a granulate, so that release of ammonium is slower. In general, ammonium concentration in runoff from the 8-metre-long fertiliser-amended plot was higher than in runoff from the 3-metre-long fertiliser-amended plot; however, there was no such consistent difference in nitrate concentrations.

Table 1. Absolute masses of ammonium and nitrate (mg) in runoff collected in each of the troughs after each simulated rainfall event, and rainfall depths (mm) applied to each subplot.

	NH_4^+			*NO_3^-*			*Rainfall (mm)*		
	Rainfall Event			*Rainfall Event*			*Rainfall Event*		
Trough	*1*	*2*	*3*	*1*	*2*	*3*	*1*	*2*	*3*
1a	79.39	0.46	0.38	2.46	1.27	1.96	23.5	43.1	45.4
1b	53.59	0.18	0.23	1.86	2.13	1.79			
1c	7.87	0.24	1.04	0.18	2.08	9.07			
1d	321.10	63.86	14.78	8.32	188.64	51.14			
2a	13.28	0.38	0.34	3.46	0.76	2.30	39.2	45.4	40.0
2b	0.43	0.53	1.54	0.74	5.52	0			
2c	47.44	1.05	0.42	3.71	33.74	0.80			
2d	0.17	0.19	0.09	1.41	1.28	0			
3a	0	0.75	0.34	3.28	0.74	1.54	53.3	62.7	92.4
3b	0	0.21	0.25	0.58	0.70	0.14			
3c	0	0.02	0.06	0.64	0.13	0.15			
3d	0	0.06	0.44	1.06	0.40	3.21			
4a	0.11	0.47	0.13	1.73	8.57	0.87	65.0	78.3	69.7
4b	0.01	0.01	0.25	0.29	0	1.14			
4c	0	0.07	0.26	1.24	1.18	4.46			
4d	0	0.11	0.10	0.92	0.78	0.82			

Estimated absolute masses of ammonium and nitrate removed in runoff are listed in Table 1. In runoff collected downslope of the slurry-amended subplots, there is a close relationship (particularly after the first simulated rainfall event) between absolute mass and runoff volume; indeed, in some cases the ranking of troughs by masses removed bears little relation to that which would be expected in view of the length of buffer strip travelled. These data are thus of little value for comparing the effectiveness of buffer strips of different length. In runoff following subsequent rainfall events, the ranking of troughs by absolute masses was different; this can be attributed to downslope redistribution of contaminants already removed from the amended plots (see Díaz-Fierros *et al.*, 1990). The absolute mass data for runoff from the fertiliser-amended subplots suggest that there was some attenuation by the longer buffer strips, though such effects were by no means consistent.

Correlations between runoff percentages and absolute masses (for each contaminant, each simulated rainfall event and each subplot type) were calculated. For runoff from slurry-amended plots, strong positive correlations were observed in all cases. For runoff from fertiliser-amended plots, correlations were strong for the first two rainfall events (between 0.994 and 0.920) but weak for the third (0.756 and 0.092). The strength of the correlations may be related a) to the extent to which runoff is able to solubilise and transport the contaminants in question, and b) to the extent to which certain runoff events provoked movement of contaminants within the buffer strip area. In this context, it is probably particularly relevant that the contaminants were applied in solid form in the fertiliser, but in dissolved form in the slurry.

Percentage attenuation of contaminant concentrations (with respect to initial concentrations in slurry) by buffer strips downslope of the slurry-amended plots in all cases exceeded 91% (except for nitrate concentration in samples from trough 1c after rainfall events 2 and 3 – 69% and 43% respectively -; these relatively high values are probably attributable to oxidation of ammonium). It should be stressed that the slurry was applied only on a single occasion, but that leaching and transport occurred during and after each rainfall event; despite this, these data clearly indicate an important attenuating effect.

Table 2. Ratios of absolute masses collected in each trough after each rainfall event to absolute masses of nitrate and ammonium applied (to the whole of that plot).

Slurry-Amended Subplots

Plot-to-Trough Distance	*Rainfall Event 1*		*Rainfall Event 2*		*Rainfall Event 3*	
	NO_3^-	NH_4^+	NO_3^-	NH_4^+	NO_3^-	NH_4^+
2 m	0.0909	0.0667	0.0313	0.0006	0.0667	0.0005
4 m	0.0400	0.0385	0.1250	0.0005	0.0278	0.0013
6 m	0.0625	0.0400	0.5000	0.0009	0.1667	0.0010
8 m	0.1429	0.2500	3.3333	0.0455	0.7692	0.0106

In the case of the fertiliser-amended plots, the contaminants were not applied in dissolved form, and it is therefore not possible to estimate percentage attenuation in the same way as for the slurry-amended plots. The ratios of masses in runoff to masses applied are very high (Table 2), but this is not an effective index of attenuation since masses collected in each trough after each rainfall event are being compared with total masses applied to the whole plot. However, ammonium concentration was consistently highest in runoff from the slurry-amended plot, whereas nitrate concentration was highest in runoff from the fertiliser-amended plot only after the first rainfall event (despite the fact that nitrate dose was much higher for the latter plot); this suggests that percentage attenuation of contaminant levels in runoff from the fertiliser-amended plot was similar to that for the slurry-amended plot.

CONCLUSIONS

Particularly after the first simulated rainfall event, runoff percentages for the slurry-amended plot were considerably higher than for the fertiliser-amended plot, suggesting that slurry application led to a temporary blockage of soil pores. Because of localised channel flow, some troughs collected a disproportionately large amount of runoff.

Despite the much higher nitrate content of the inorganic fertiliser, both nitrate and ammonium levels were consistently higher in runoff from the slurry-amended plot than in runoff from the fertiliser-amended plot. The only major exception was after the first simulated rainfall event, when nitrate levels were higher in runoff from the fertiliser-amended plot. These results indicate that the risk of pollution events due to transport of ammonium and nitrate in runoff is higher after application of slurry than after application of inorganic fertiliser. The data on ammonium and nitrate levels in runoff collected downslope of the slurry-amended plot indicate that buffer strips are effective for reducing levels of these contaminants, with percentage attenuations greater than 91% in all cases except for nitrate after the second and third rainfall events. However, the percentage attenuation estimates provide only a rough indication of real effectiveness, since the slurry was applied on a single occasion, whereas leaching and dilution took place after each of the three simulated rainfall events. Strong correlations were observed between runoff percentages and masses of contaminants in runoff.

The results of this study thus suggest that buffer strips are effective for reducing ammonium and nitrate levels in runoff from slurry- or fertiliser-amended agricultural land. However, there can be no guarantee that buffer strips will reduce the concentrations of such contaminants to acceptable levels, particularly if localised channel flow occurs.

REFERENCES

APHA. (1989) *Standard Methods for the Examination of Water and Wastewater.* 17th ed., American Public Health Association, Washington.

Basanta, R., Núñez, A., López, E., Fernández, M. and Díaz-Fierros, F. (1995) Measurement of cholinesterase activity inhibition for the detection of organophosphorus and carbamate pesticides in water. *Intern. J. Environmental Studies,* **48**, 211-219.

Bremner, J.M. and Keeney, D.R. (1965) Steam distillation methods for determination of ammonium, nitrate and nitrite. *Anal. Chim. Acta,* **32**, 485-495.

Carballas, T., Díaz-Fierros, F., Acea, M.J., Cabaneiro, A., Carballas, M., Gil, F., Leirós, M.C., López, E., Núñez, A., and Villar, M.C. (1990) *El Purín de Vacuno en Galicia. Caracterización, Poder Fertilizante y Problemas Ambientales.* COTOP- Xunta de Galicia, Santiago de Compostela (Spain).

Castilho, P., Chardon, W.J. and Salomons, W. (1993) Influence of cattle-manure slurry application on the solubility of cadmium, copper and zinc in a manured acidic, loamy-sand soil. *Journal of Environmental Quality,* **22**, 689-697.

Coyne, M.S., Gilfillen, R.A., Rhodes, R.W. and Blevis, R.L. (1995) Soil and fecal coliform trapping by grass filter strips during simulated rain. *J. Soil Water Cons.* **50**(4), 405-408.

Díaz-Fierros, F., Núñez, A. and López, E. (1990) Risks of water pollution by superficial run-off resulting from the spreading of cattle slurry. In: Calvet, R. (Ed.) *Nitrates, Agriculture, Eau* . INRA Editions, Paris, 481-486.

Díaz-Fierros, F., Núñez, A. and López, E. (1993) *As concas fluviais de Galicia. Características e riscos de contaminación difusa.* Univ. de Santiago. Servicio de Publicacións e Intercambio Científico, Santiago de Compostela.

Dickey, E.C. and Vanderholm, D.H. (1981) Performance and design of vegetative filters for feedlot runoff treatment. In: *Livestock Waste: A Renewable Resource,* American Society of Agricultural Engineers, 257-260.

Dillaha, T.A., Sherrard, J.H., Lee, D., Shanholtz, V.O., Mostaghimi, S. and Magette, W.L. (1986) Use of vegetative filter strips to minimize sediment and phosphorus losses from feedlots: Phase I. Experimental plot studies. *Virginia Water Resources Research Center Bulletin* **151**, 68.

Núñez, A., López, E. and Díaz-Fierros, F. (1991) Contaminación das augas por escorremento superficial en pradeiras en pendente tratadas con purín de vacuno. *Cuad. Area Cienc. Agr.,* **11**, 163-182.

Núñez Delgado, A. (1993). *Riesgos de contaminación de las aguas continentales por aplicación de purín sobre praderas en pendiente.* PhD Thesis. Univ. Santiago, Santiago de Compostela.

Núñez, A., López, E. and Díaz-Fierros, F. (1995) Vegetated filter strips for wastewater purification: a review. *Bioresource Technology,* **51**, 13-22.

Scow, K.M., Fan, S., Johnson, C. and Ma, G.M. (1995) Biodegradation of sorbed chemicals in soil. *Environ. Health Persp.,* **103**(5), 93-95.

15 The Potential Role of Ponds as Buffer Zones

S. Fleischer, A. Joelsson and L. Stibe

County Administrative Board, S 301 86 Halmstad, Sweden

Abstract

In many countries eutrophication of coastal waters has increased. Governmental programmes and international agreements to counteract eutrophication have largely not attained agreed goals (e. g. reduction by half of the anthropogenic nitrogen load on Swedish coastal waters, to be carried out between 1985 and 1995). Important building blocks in such programmes are improved removal of nitrogen in municipal treatment plants and agricultural measures, e.g. use of cover crops or improved manure management. Agricultural measures have shown efficient reductions of nitrogen leaching under experimental conditions, but have so far not resulted in decreased nitrogen transport from farmland.

In an interdisciplinary project in the Laholm Bay drainage basin (SE Kattegat) a package of measures were identified to decrease nitrogen transport to the coast. One of these measures did not decrease leaching or emissions, but increased removal during runoff, i.e. restoration of ponds and wetlands. Budget studies in existing ponds/wetlands showed a relationship between the areal nitrogen load and the areal nitrogen removal. Budget studies in full scale restored ponds later verified this general view. Per area unit, increased nitrogen loading implied increased nitrogen retention, but often decreased percent retention. In this project ponds with depths 0.4-2.0 m and hydrological loads 0.14-5.2 $m^3 m^{-2} day^{-1}$ were created. 150-7000 kg N $ha^{-1} yr^{-1}$ and 18-404 kg P $ha^{-1} yr^{-1}$ was removed in ponds loaded by streams. A pond receiving pre-treated municipal wastewater removed 8000 kg N $ha^{-1} yr^{-1}$ and 590 kg P $ha^{-1} yr^{-1}$. The upper limit for N-removal is set by the hydrological conditions. Sedimentation of organic material must be favoured in order to obtain adequate conditions for denitrification at the sediment-water interface. In the long run, channelisation should be avoided by appropriate management. High loaded ponds are cost efficient, as they reduce much nitrogen in small created units, each with low costs.

Creation/restoration of ponds and wetlands has now become part of the Swedish agri-environmental programme. One problem is, however, that ponds should be localised to strategic sites in the catchments, rather than to sites pointed out by the farmers. An inventory of possible sites for pond/wetland creation was made for the region, based upon air photography, and the effects of pond/wetland constructions were computed in small drainage areas.

To attain the agreed goal of a 50 percent reduction of the nitrogen transport in streams, decreased agricultural leaching must be combined with extensive pond and wetland construction.

INTRODUCTION

Degradation of wetlands

Construction of dams, channels and large scale drainage of both agricultural and forested land has completely changed many water systems. In some areas in southern Sweden, the area of wetlands has been decreased considerably (more than 90%) by drainage. The degradation of rivers, streams and wetlands has become of increasing concern, not only to ecologists and limnologists but also to the general public. However in recent years, water managers have become more aware of the natural processes in watercourses which are vital for the long-term health of the environment.

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0

The condition of wetlands and riparian zones is closely linked to the development of agriculture and forestry. Up to the 1970s, streams were channelised and deepened, and wet areas drained for agricultural purposes (Fleischer *et al .*, 1987). In forested land, bogs and swamps were to a large extent drained until the late 1980s (Fleischer & Stibe, 1989). Today, attitudes to drainage are more restrictive at the official level, but drainage is also looked upon more sceptically by foresters and farmers.

Increasing nutrient loading

The SE Kattegat, a shallow estuarine area which connects the Baltic Sea with the Atlantic Ocean, has had eutrophication problems due to nitrogen overloading since the 1970s. In the Laholm Bay drainage basin, with about 12% arable land, agriculture is responsible for nearly half of the nitrogen load to streams (Fleischer and Jonsson, 1992). The coastal plain is one of the most intensively farmed areas in Sweden. There is a predominance of livestock farms, in an area which has mostly sandy soils which are prone to nutrient leaching. The large losses of nitrogen to streams (40-50 kg N ha^{-1} is common) along the coastal strip are aggravated by intensively cultivated arable fields, bare soils in winter, and a relatively wet and mild climate.

In long term field studies, made within the catchment since 1984, Lindén *et al.* (1993) and Torstensson *et al.* (1994), have shown the influence of manure application, ploughing time and an undersown catch crop on N-mineralisation and leaching. The undersown catch crop (ryegrass) appeared to be the most promising way to decrease nitrogen leaching from arable land. Perennial ryegrass, undersown in spring cereals, decreased nitrogen leaching to one third in comparison with a bare soil in winter.

In 1986 an action programme was set up in order to reduce nutrient leaching from the drainage basin, and from agriculture in particular. The measures, based mainly on the results above, included field control and monitoring of manure storage, techniques for spreading of manure and fertilisers, wintergreen crops and undersown catch crops, as well as treatment of sewage from urban and rural areas. Special guidelines, extension services and state subsidies have been introduced to help farmers avoid water pollution.

Despite this programme and considerable investments in improved sewage treatment, increased manure storage capacity and the strict regulation of agricultural practices, there has been no decrease in nutrient transport into the sea over this period. This may partly be explained by the milder and wetter weather experienced during the winter months, and the increased contribution from forested areas. Future strategies should involve targeting catchments where nitrogen losses are largest and identifying sources within these catchments. The creation of wetlands in agricultural areas has also been given a high priority. The present agri-environmental programme of the EU includes restoration of wetlands (compensation US$ 685 ha^{-1} and per year). According to topographical studies it has been estimated that up to 1,200 potential sites exist for the creation or restoration of wetlands in the County of Halland (Wessling, 1991).

CREATION OF PONDS TO IMPROVE WATER QUALITY

An empirical relationship between the areal loading and retention of nitrogen (kg N.ha^{-1} yr^{-1}) in ponds, including both denitrification and accumulation in plant material and sediment, was compiled from budget studies and literature (Fleischer and Stibe, 1991) and verified within the "Halmstad project" (Fleischer *et al.*, 1994), where ponds/wetlands with depths between 0.4 and 2.0 m were created. The objective was to determine the loading limits and emissions of nitrogen gases, to improve our understanding of the retention processes and present a prerequisite for cost effective water management. The study was carried out within a programme facing the coastal water eutrophication problem initially described, and showed that periods of retention were frequently followed by periods of release of nitrogen. However the annual net retention was estimated at 73-7000 kg N ha^{-1} in the created wetlands in agricultural streams and urban stormwater drains (Fleischer *et al.*, 1994). About 8000 kg N ha^{-1} and per year was retained in a pond receiving pre-treated wastewater. Jacks *et al.* (1994)

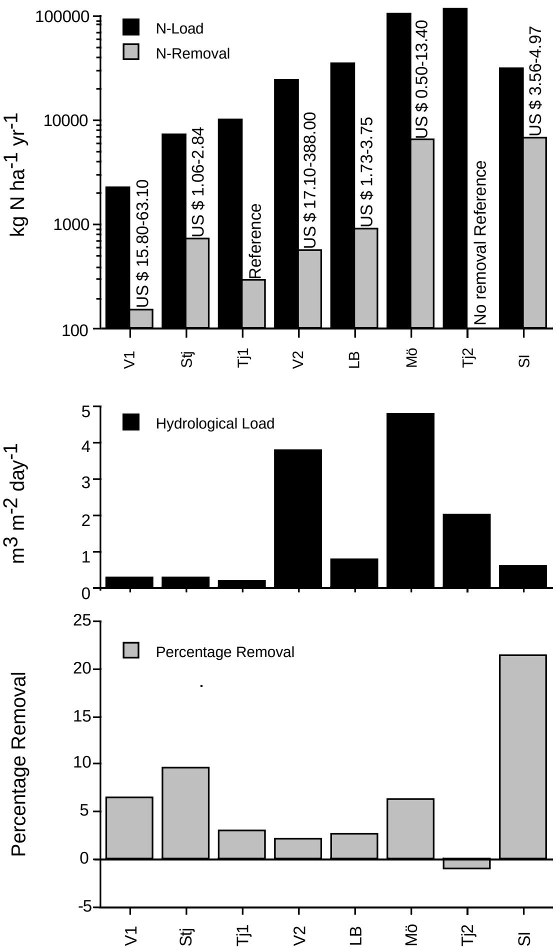
have shown that forest wetlands in southern Sweden, receiving a large nitrogen load, also act as nitrogen traps. Considerable nitrogen retention occurs in a forest wetland where nitrogen load is increased as a result of disturbances such as clear cutting and scarification, upslope.

Nitrogen

An increased N load per unit area caused an increased areal removal (retention) in the created ponds, but a decreased percentage removal; the hypothesised dose/response relationship was verified by Fleischer *et al.* (1994) and Fleischer (1995a). The capacity for retention is limited by extreme hydrological loadings, which prevent sedimentation. Sedimentation creates a sediment rich in organic material, which is a pre-requisite for denitrification. On the other hand, too large an accumulation of organic material in a stagnant pond may result in conditions too reduced for optimal denitrification (Fleischer, 1995b).

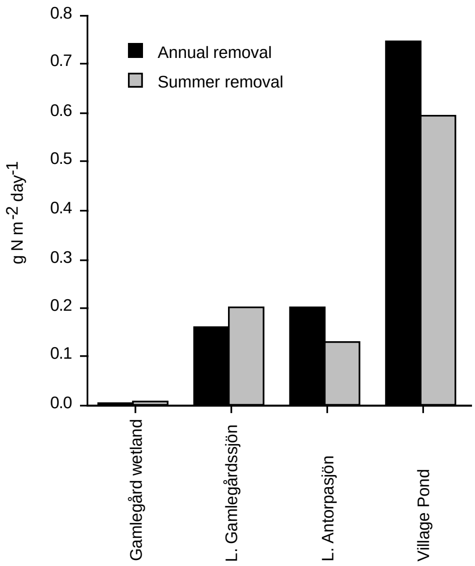
Measurements of retention capacity also made it possible to estimate marginal costs for N reductions. Despite the large cost of creating the Möllegård pond (US$ 85000 for 1.3 ha), because of the use of excavators, the marginal cost became very low, as a result of the large areal N retention in the pond. The calculations considered investment (an interest rate of 10% was assumed), opportunity and maintenance costs, but not inflation (Fleischer *et al.* 1994). The minimum and maximum length of life was also estimated, and therefore the marginal cost for each pond varied within a range. The results are summarised in Fig. 1.

Figure 1. Nitrogen load/removal and marginal costs ($ per kg N) in six created ponds (Stj, LB and Mö = runoff from agricultural land, V1 and V2 = urban stormwater, Sl = treated wastewater) and two reference ponds (TJ1 and TJ 2). In the lower part of the figure the hydrological load (left axis) and percent removal (right axis) are shown. Results from a 12-month budget study in SW Sweden.



The efficiency of ponds/wetlands at removing nitrogen during wintertime is sometimes questioned, because biological retention processes (assimilation, denitrification) are slow at low temperatures. However, as a result of the large areal N load during wintertime, areal N removal is at the same level as during summer, when nitrogen removal sometimes is limited by the small N input (Fig. 2).

Figure 2. Annual and summer nitrogen removal in two lakes, one wetland and one village pond studied during the same time period.



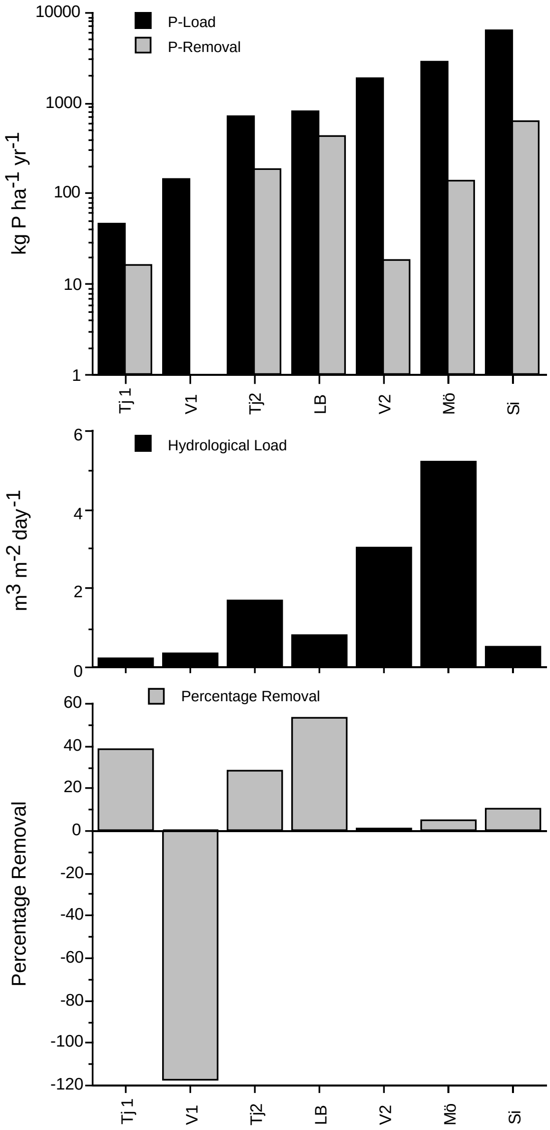
Phosphorus

Phosphorus removal was also studied in the created ponds by means of budget calculations. The results varied widely, and no clear-cut relationship between P and N retention was found (Pansar and Stibe, in print). This is logical when considered against the different types of P removal processes in the limnic environment (Boström *et al.,* 1988). A large annual total-P removal (1200 kg ha^{-1} initially, decreasing to 590, mean = 780) occurred in municipal wastewater, pre-treated by 8 hours aeration (Pansar *et al.,* 1996). This activated sludge pre-treatment, which fulfilled the environmental demands for a local wastewater outlet from a village, also converted nitrogen to a form available for denitrification (nitrate) and, at the same time, created conditions for efficient P removal.

Large P losses to downstream waters occurred from one pond receiving municipal stormwater (V1 in Fig. 3). The average hydrological load was rather low in this shallow pond (0.30 m^3 m^{-2} day^{-1}), but increased incidentally during storm events, causing erosion losses of P. P losses also resulted from the large bird populations in this pond (Pansar and Stibe, in print). The birds were fed by people living in the surrounding area, a P input that was difficult to estimate. The birds also contributed to more reduced conditions at the sediment surface, with decreased capacity for P removal as a result.

Despite the almost three times larger hydrological load in the agricultural pond (LB in Fig. 3), this pond showed efficient P removal. This was largely due to sedimentation during periods of high discharge carrying large amounts of particulate P, and also occurred in other ponds (Pansar and Stibe, in print). Conversely, during periods of low discharge and low transport of particulates, areal P removal was less effective. This is probably an effect of decreasing redox potential at the sediment/water interface as a result of stagnant water. Conditions which are too reduced are apparently unfavourable for both N and P removal.

Figure 3. Phosphorus load/removal, hydrological load and percent removal in the same ponds (except Stj) and during the same period as in fig. 1.



THE POTENTIAL ROLE OF PONDS IN WATER MANAGEMENT: CASE STUDIES

N and P have been deposited in sediments since the last glaciation. This deposition occurs together with organic material and does not result in "nutrient saturation". In addition, NO_3 is also converted to N_2 in sediments as a result of denitrification. Release of N_2O is not a problem, as this greenhouse gas is produced predominantly in the catchment soils (Fleischer *et al.*, 1994, Fleischer and Pansar, in print). Therefore, in the long term, water bodies are nutrient sinks, and the same removal processes are the basis for use of ponds and wetlands in water management. Even if some CH_4 is produced in the wetlands, their overall role seems to be positive, as the net effect of buffer zones is decreased, coastal eutrophication with less heterotrophic conditions (Smith and Mackenzie, 1987) and thus, decreased release of greenhouse gases from the sea.

However strong, the nitrogen load/retention relationship should not be used to predict nitrogen retention in single ponds or wetlands, but merely as a tool to estimate the results of large scale wetland creation or restoration. The potential of ponds was earlier demonstrated for the River

Genevadsån drainage basin, SW Sweden (Stibe, 1991, Fleischer and Stibe, 1994), where a 60-65 ton reduction (20%) of the nitrogen transported was calculated. This study also confirmed our hypotheses regarding ponds and wetlands as a cost effective complement to traditional measures for nitrogen reduction.

In a new project describing and quantifying the environmental improvements that can be established on a voluntary basis, through the adjustment of the agricultural practices, the potential of ponds has been analysed. The studied drainage area of River Nyrebäcken covers 51 km^2 and is dominated by agricultural land (70%). A large part of the agricultural area is tile drained and many former streams are today converted to culverts. The present nitrogen transport amounts to approximately 90 tons per year. Average nitrogen losses from arable land are estimated at 25 kg ha^{-1} as an average for the total area.

An inventory in the River Nyrebäcken drainage area, based on aerial photography, identified 11 potential wetlands or ponds which could be created by damming (Wessling, 1991). In several of these wetlands, nitrogen retention would probably be limited by too large a hydrological load, and some are situated in very small tributaries where nitrogen retention would have only marginal effect compared to the total river transport. However, in the lower part of the river, three potential ponds in a series were identified. By extended damming, a large wetland of approximately 10 hectares could be created, with a relatively low hydrological load, and sufficient turn over.

In addition, many existing village ponds (resulting from farmers digging for marl in the 19th century) could also be used. Typically these ponds are isolated, with neither inlet nor outlet. Leading the culverted ditches to these ponds would also decrease the agricultural nitrogen losses to the main river.

Assuming that 50 of the total 100 marl ponds are used, and that the runoff from half the agricultural area is distributed to these ponds, a total nitrogen removal of 12 tons could be achieved according to the load/retention relationship. In combination with the 10 ha in-stream wetland, nitrogen transport in the stream is predicted to decrease by a total of 33 tons (36%).

CONCLUSION

As a part of a catchment based action programme to improve water quality, nutrient retention ponds apparently constitute an attractive option. The marginal costs of nitrogen and phosphorus retention in created ponds are low, which should be considered in water management programmes. Both measures to decrease nitrogen leaching and to increase nitrogen removal are necessary in many areas. In most created ponds and wetlands phosphorus removal and increased biodiversity are additional benefits.

REFERENCES

Boström, B., Andersen, J. M., Fleischer, S. and Jansson, M. (1988) Exchange of phosphorus across the sediment-water interface, *Hydrobiologia,* **170**, 229-244.

Fleischer, S. (1995a) Changed production and composition of sedimentary gases as a response to increased nitrogen load in lakes and wetlands, *Annali di Chimic,* **85**. Societa Chimica Italiana.

Fleischer, S. (1995b) Management of nitrogen in drainage basins – Laholm Bay, Sweden. In: Eiseltova, M. and Biggs, J. (Eds.) *Restoration of Stream Ecosystems. An integrated catchment approach,* IWRB Publication 37.

Fleischer, S. Hamrin, S. Kindt, T. Rydberg, L and Stibe, L. (1987) Coastal Eutrophication in Sweden. Reducing Nitrogen in Land Runoff, *Ambio,* **16**(5), 246-251.

Fleischer, S and Stibe, L. (1991) Drainage basin management – reducing river transported nitrogen, *Verhandl. Internat. Verein. Limnol.,* **24**, 1753-1755.

Fleischer, S. Gustafson, A. Joelsson, A. Pansar J. and Stibe L. (1994) Nitrogen removal in created ponds, *Ambio,* **23**(6), 349-357.

Fleischer, S. Jonsson P. (1992) *Measure counteracting eutrophication of the Kattegat,* Science of the Total Environment, Supplement 1992.

Fleischer, S. and Stibe, L. (1989) Agriculture kills marine fish in the 1980's. Who is responsible for fish kills in the year 2000? *Ambio,* **18**(6), 346-349.

Fleischer, S. and Stibe, L. (1994) Decreasing nitrogen load on coastal waters. Interdisciplinary approach stresses the need for wetland restoration, *Verhandl. Internat. Verein. Limnol.*, **25**, 1898-1900.

Fleischer, S. and Pansar, J. (In print) N_2O transport from agricultural and forest soils to the drainage water, *Vatten.* In Swedish, English summary.

Jacks, G., Joelsson, A. and Fleischer, S. (1994) Nitrogen retention in forest wetlands, *Ambio,* **23**, 358-362.

Lindén, B. Gustafson, A. and Torstensson, G. (1993) Influence of pig slurry, ploughing time and a ryegrass catch crop on N-mineralization and leaching, Proceedings of NJF-seminar no. 228, *Soil Tillage and Environment.* Jokionen, Finland.

Pansar, J., Fleischer, S., Leinenbach, P., Ohlsson, L. and Lundin, M. (1996) Ponds as a final step in wastewater treatment – nutrient reduction. In: Vymazal, J. (Ed.) *Nutrient Cycling and Retention in Wetlands and Their Use for Wastewater Treatment. Ecology and Use of Wetlands,* Prague and Botanical Institute, Trebon.

Pansar, J. and Stibe, L. (In print) Phosphorus retention in created wetlands designed for nitrogen removal, *Verhandl. Internat. Verein. Limnol.,* **26**.

Smith, S. V. and Mackenzie, F. T. (1987) The ocean as a heterotrophic system: Implications from the carbon biochemical cycle, *Global Biochemical Cycle,* **1**, 187-198.

Stibe, L. (1991) Wetlands as nitrogen sinks – Towards a model for planning of wetland restoration, *Vatten,* **47**, 286-290 (in Swedish summary in English).

Torstensson, G. Aronsson, H and Lindén, B. (1994) *Winter crops as green cover crops – nitrogen uptake capacity and effects on nitrogen leaching.,* Proceedings of NJF-seminar no. 245. Swedish University of Agricultural Sciences.

Wessling, M. (1991) *Potential wetlands in the county of Halland. Part 2. Data from an inventory of potential wetlands in the agricultural landscapes of Halland.* – Swedish University of Agricultural Sciences, Research report 91:7 (in Swedish).

16

The potential role of wet meadows and grey alder forests as buffer zones

Ü. Mander[1], K. Lõhmus[1], V. Kuusemets[1] and M. Ivask[2]

[1] *Institute of Geography, University of Tartu, 46 Vanemuise St., EE2400 Tartu, Estonia*
[2] *Environmental Protection Institute, Estonian Agricultural University, 5 Kreutzwaldi St., EE-2400 Tartu, Estonia*

Abstract

Two differently loaded riparian buffers including wet meadows *(Filipendula ulmaria-Aegopodium podagraria – Cirsium oleraceum – Urtica dioica)* and grey alder *(Alnus incana)* stands were investigated. The main inputs and outputs (atmospheric deposition, input and output in surface and subsurface flow, nitrogen fixation and denitrification) and accumulation of N and P in plant biomass, litter and soil were estimated. A 31 m wide buffer zone of wet meadow and grey alder forest removed 67% nitrogen and 81% phosphorus, while in a 51 m buffer zone, also containing a grassland strip in addition to wet meadow and alder forest, 96% N and 97% P was retained. In the riparian buffers studied, an effective retention of nutrients (34-186 kg N ha^{-1} yr^{-1} and 1.6-6.0 kg P ha^{-1} yr^{-1}) was observed even with very high input loads (276.3 and 12.8 kg ha^{-1} yr^{-1} of N and P, respectively).

INTRODUCTION

Riparian wet meadows and forests have been identified as important buffers for water bodies (Lowrance *et al.,* 1984; Peterjohn and Correll, 1984; Haycock and Pinay, 1993; Vought *et al.,* 1994). Most research only considers inputs and outputs, but among the internal processes of riparian ecosystems relevant to nutrient retention, denitrification has been investigated most intensively (Groffmann *et al.,* 1991; Lowrance, 1992; Weller *et al.,* 1994; Pinay *et al.,* 1993). Only a few studies deal with plant uptake and soil accumulation within riparian forests (Peterjohn and Correll, 1984; Lowrance *et al.,* 1984) and wet meadows (Leonardson *et al.,* 1994). The aim of this paper is to analyse the most important nutrient fluxes and pools in complex riparian buffer zones to explain their transformation and removal efficiency in differently loaded areas.

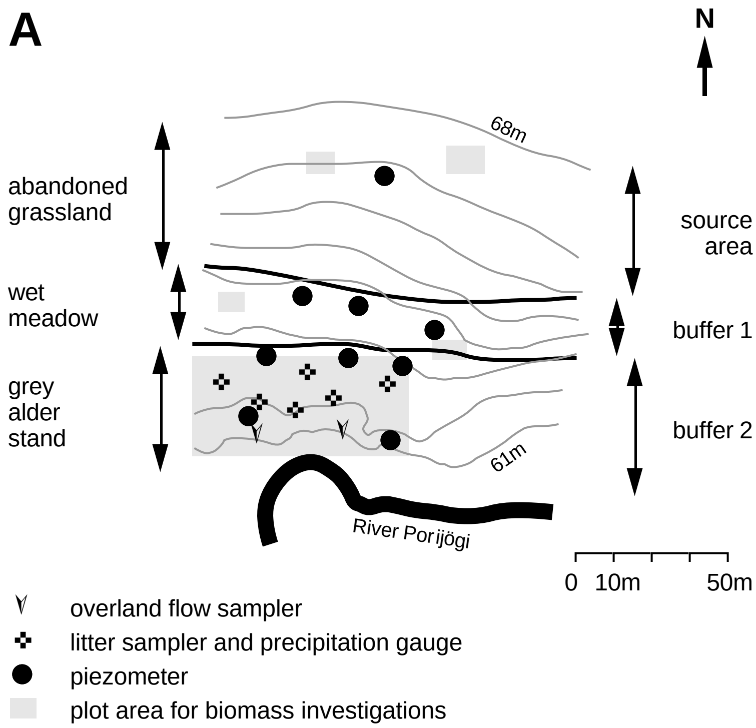
SITE DESCRIPTION

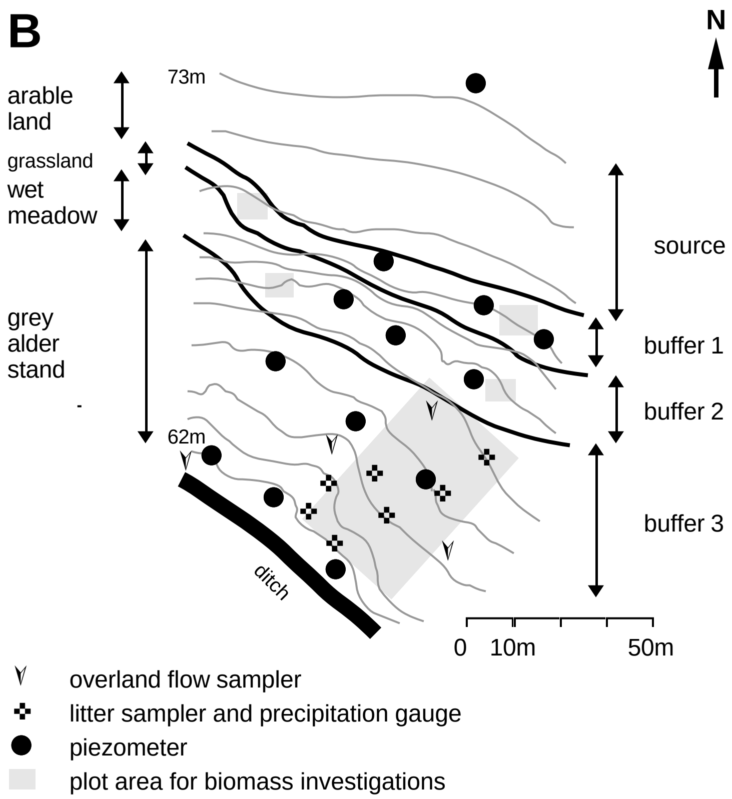
Two different riparian buffers including wet meadows and grey alder stands were selected in Estonia: one in the unpolluted Porijõgi River catchment (for area description see Mander *et al.,* 1995), the other, in the vicinity of the Viiratsi pig farm (32,000 pigs), Viljandi County. The physio-geographical conditions of the Viiratsi study site are similar to those of the Porijõgi River catchment. In both areas, transects in thalwegs were established along a topo-edaphic gradient in autumn 1993 (Fig. 1). In the Porijõgi test site the following series of riparian buffer communities, in the downhill direction, was analysed: abandoned (formerly cultivated) grassland (serves as source community) – buffer 1; wet meadow (dominated by *Filipendula ulmaria, Aegopodium podagraria, Cirsium oleraceum,* and *Urtica dioica),* 11 m – buffer 2; grey alder stand (14 yr), 20 m. In the heavily polluted Viiratsi test site the transect was established through the following communities: arable land (fertilised by pig slurry; serves as source community) – buffer 1; eutrophic grassland strip *(Elytrigia repens, Urtica dioica)* with a young grey alder stand, 11 m – buffer 2; wet meadow pattern *(Filipendula ulmaria),* 12 m – buffer 3; grey alder forest (40 yr), 28 m. In the landscape profiles piezometers (3 rows in the Porijõgi transect and 4 rows in the Viiratsi site, with 3 replicates in each row) and study plots were established on the boundaries between plant communities. The main nitrogen (N) and phosphorus (P) cycles and

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0 © 1997 Quest Environmental.

budgets have been assessed. This paper presents the main inputs and outputs (atmospheric deposition, input and output in surface and subsurface flow, nitrogen fixation and denitrification) as well as accumulation in plant biomass, litter and soil from July 1994 to July 1995.

Figure 1. Schemes of test sites: A – Porijõgi (less polluted), B – Viiratsi (heavily polluted).





MATERIAL AND METHODS

Productivity and uptake estimation

Dimension-analysis techniques (Bormann and Gordon, 1984; Rytter, 1989; Huss-Danell and Ohlsson, 1992) were used to estimate the above-ground biomass and productivity of grey alder forests. At both test sites (age 14 and 40 years) 17 and 5 model trees per plot, respectively, were felled to collect data on the following tree components: stem (wood and bark), secondary branch growth (wood and bark), primary branch growth, leaves, generative organs. The relative increments of the wood and bark of an overbark fraction were assumed to be equal. Root systems for 6 and 3 out of the sampled 17 and 5 trees respectively were excavated and divided into five fractions; stump, coarse roots: $d \geq 20$ mm, 5 mm $\leq d < 20$ mm, 2 mm $\leq d < 5$ mm and fine roots ($d < 2$ mm). Nodule mass (kg ha^{-1}) was estimated separately in June and July 1995. To estimate the below-ground production, the shoot/root ratios for tree biomass and production were assumed to be equal.

All tree components were analysed for N, P, energy and ash contents. Tree components have been estimated using the regression equation:

$$\ln y = a + b \ln dbh \qquad (1)$$

where y is the oven-dry mass of tree component (kg) and dbh – diameter at breast height (cm); all equations had very high correlation coefficients and significance ($p < 0.0001$ in all cases; Table 1).

Table 1. Parameters of regression equations (1) used in dimension analysis for estimating the mass of tree compartments (kg); r^2 – coefficient of determination, s.e.e. – standard error of estimate.

Age (years)	*Tree compartment overbark (kg)*	*a*	*b*	*r^2*	*s.e.e.*
14 (Porijõgi)	Stem	-2.492	2.399	0.992	0.07
	Branches	-6.064	3.123	0.925	0.31
40 (Viiratsi)	Stem	-2.406	2.354	0.984	0.14
	Branches	-3.891	2.353	0.947	0.33

The phytomass (*i.e.*, standing crop) samples were collected during the maximum flowering time of the dominant plant species (2nd and 3rd week in July; see Milner and Hughes, 1968) from all riparian plant communities. Sampling plots (six in Porijõgi and three in Viiratsi), for analysing plant cover and phytomass, were installed in typical areas of the community. Two typical patches within the wet meadow community (dominated by *Aegopodium podagraria* and *Filipendula ulmaria*) in the Porijõgi transect were analysed (see Balsberg, 1982). Above-ground biomass was collected from three replicate quadrats (1x1 m) in each community. Below-ground root biomass was collected from soil cores taken by auger (diameter 158 mm) from the depth of up to 40-50 cm in three replicates from each location.

Field experiments and laboratory analysis

Water samples were collected and groundwater depth measured once or twice a month by piezometers. Filtered soil water samples were analysed for NH_4-N, NO_2-N, NO_3-N, total Kjeldahl nitrogen (TKN), PO_4-P, total Kjeldahl phosphorus (TKP), SO_4, Fe, Ca (APHA, 1989). Soil bulk density, texture class and field capacity were determined for each 20 cm of soil profile (up to 1.5 m depth). Hydraulic conductivity was estimated by using tracer (chloride) and pumping experiments (Freeze and Cherry, 1979). Groundwater discharge was estimated on the basis of both Darcy's law and by gauging with weirs installed in groundwater seeping sites. TKN and TKP of plant samples were estimated.

The acetylene method was used to measure the N fixation rate in soil (Groffmann and Tiedje, 1989). To assess the denitrification rate acetylene as a nitrous oxide reduction inhibitor was used (Yoshinari *et al.*, 1977).

RESULTS AND DISCUSSION

Nitrogen and phosphorus budgets in differently loaded buffer zones

Nitrogen

Despite the significantly higher N loading in the Viiratsi riparian buffer zone relative to the Porijõgi complex (0.4-4.3 and 2.0-62.1 mg N l^{-1}, respectively) the output concentrations were comparable (0.4-2.1 and 0.5-3.0 mg N l^{-1}). Atmospheric N deposition in Porijõgi and Viiratsi was estimated to be 6.1 and 6.3 kg N ha^{-1} yr^{-1}, respectively. Extremely high TKN contents in the Viiratsi soil water (56.1 and 62.1 mg l^{-1}) reflect pig slurry application in the adjacent field in July and August 1994. Intensive fertilisation over many years has compacted the soil and disturbed the microfauna. Therefore the N concentration in soil water under the arable land has always been high. In the Porijõgi catchment, by contrast, the N input has decreased during the last three years since agricultural activities in the upland field ceased. The high buffering capacity in the Viiratsi study (Table 3) is the result of: (1) the large accumulation of organic nitrogen in the soil, (2) the relatively high plant uptake, (3) the relatively high denitrification value, (4) the relatively low N_2 fixation. Fluxes (2)-(4) were generally smaller than those in the Porijõgi test site (Table 2).

Table 2. Main fluxes and pools of nitrogen and phosphorus in the heavily polluted test site at Viiratsi (kg ha^{-1} yr^{-1}).

Fluxes and pools	*Grassland*		*Wet meadow*		*Alder forest*	
	N	*P*	*N*	*P*	*N*	*P*
Precipitation	6.3	3.1	6.3	3.1	6.7	1.0
Nitrogen fixation	5.5		0.8		21.0	
Input surface flow and subsurface flow	264.5	9.7	79.0	3.7	45.2	2.0
Accumulation in plant biomass	165.0	45.0	352.0	38.0	140.2	10.8
Litter	*78.2	*21.9	*201.8	*17.9	87.0	4.0
Denitrification	20.1		10.2		7.9	
Output surface flow and subsurface flow	79.0	3.7	45.2	2.0	9.0	0.4
Active soil exchange	90.2	-14.0	-119.5	-15.3	2.8	-4.2
Soil store (kg ha^{-1})	11.2	2.7	17.3	1.4	20.6	2.1

* above-ground litter, estimated values

Denitrification within the grey alder forest was 12-21 μg N m^{-2} hr^{-1} at Porijõgi and 3-14 μg N m^{-2} hr^{-1} in Viiratsi. Nevertheless, in adjacent wet meadow and abandoned grassland upslope from the forest in Porijõgi, the rate was higher (4-57 and 5-41 μg N m^{-2} hr^{-1}, respectively). The main characteristics in denitrification intensity are comparable with other investigations:

(a) most denitrifying activity was observed in spring and late summer (Struwe and Kjøller, 1990; Weller *et al.*, 1994)

(b) denitrification was faster in the upper part of the complex buffer zone (upslope from the alder forest; see also Duff and Triska, 1990; Pinay *et al.*, 1993; Weller *et al.*, 1994)

Characteristic (b) is due to significantly higher nitrate concentrations in soil water upslope. Struwe and Kjøller (1991) have found up to 100 times more denitrifying activity in slurry incubations than in black alder forest in the field.

Nitrogen uptake by grey alders is high at both sites. In Viiratsi the summary uptake is about 30% less than in Porijõgi, i.e. 140.2 and 204.8 kg N ha^{-1} yr^{-1}, respectively (Tables 2 and 3). The forest stand in Viiratsi is less dense and older than that in Porijõgi (1810 trees ha^{-1} by average age of 40 yr and 6110 trees ha^{-1} by 14 yr, respectively). In addition, only a portion of trees are grey alders; in Viiratsi 77% and in Porijõgi 86%. The N allocation in alder production indicates that most of the N accumulates in leaves (88.5 kg N ha^{-1} yr^{-1} in Viiratsi and 85.8 kg N ha^{-1} yr^{-1} in Porijõgi, in percentages of the total N uptake: 63% and 42%, respectively). We measured a relatively large amount of N in the bark of stems and branches (8.5 kg N ha^{-1} yr^{-1} in Viiratsi and 23.1 kg N ha^{-1} yr^{-1} in Porijõgi). Because the growth of trees in older stands is less than that in younger stands, the N uptake by stems and branches of old trees is lower than that of younger trees. Moreover, the annual N uptake stored in root production decreases in older stands. Due to slow retranslocation into senescing leaves in autumn (8% in Viiratsi and 14% in Porijõgi), most N is accumulated in leaf litter, half of which mineralises during the next season. Thus, a realistic N removal rate by tree uptake is about 40-50 kg less than the total annual uptake.

Atmospheric N_2 fixation in the alder stand in Viiratsi was significantly less than that in Porijõgi (i.e., 0.2-2.8 and 0.6-15 μg N m^{-2} hr^{-1}, respectively) with a maximum in July. The highest N_2 fixation values (up to 21.3 μg N m^{-2} hr^{-1} in Porijõgi and 17.9 μg N m^{-2} hr^{-1} in Viiratsi) were observed within wet meadows and grassland communities; most fixation was observed in May. The smaller N_2 fixation in Viiratsi is due to predominating N assimilation over N_2 fixation while high concentrations of mineral N are present in the root (see Troelstra *et al.*, 1992). However, our investigations show that N_2 fixation plays a less significant role in the total N budget in both study plots.

The active soil exchange (ASE) is calculated as follows (Tables 2 and 3):

$$\text{ASE} = \text{Input} - \text{Output} - \text{Accumulation in plant biomass} + \text{Litter} \qquad (2)$$

Except for the alder stand in Viiratsi, the active soil exchange of N for riparian communities was negative. Therefore, plant uptake exceeded the accumulation in soil during the study period. In comparison with the soil store of N in different communities (8-19 t ha^{-1} in Porijõgi and 11-21 t ha^{-1} in Viiratsi; Tables 2 and 3) the plant uptake and all other fluxes are small. Thus, the buffering capacity of colluvial soils with a deep humus layer, typical of riparian soils of agricultural areas, is the key factor in nutrient retention in studied buffer zones.

Table 3. Main fluxes and pools of nitrogen and phosphorus in the less polluted test site at Porijõgi (kg ha^{-1} yr^{-1}).

Fluxes and pools	*Wet meadow*		*Alder forest*	
	N	*P*	*N*	*P*
Precipitation	6.1	3.9	6.4	0.7
Nitrogen fixation	6.5		36.0	
Input surface flow and subsurface flow	40.0	2.5	25.6	1.8
Accumulation in plant biomass	223.0	27.0	204.8	15.1
Litter	*125.4	*12.7	82.0	4.0
Denitrification	19.3		8.5	
Output surface flow and subsurface flow	25.6	1.8	13.2	0.6
Active soil exchange	-89.9	-9.7	-76.5	-9.2
Soil store (kg ha^{-1})	7.9	1.9	19.2	19.6

* above-ground litter

Phosphorus

The P output concentration from the intensively loaded Viiratsi site is not significantly higher than that in Porijõgi, varying from 0.2 to 0.55 mg P l^{-1} and 0.08 to 0.65 mg P l^{-1}, respectively. Nevertheless, the input P values at the border of the arable land and the eutrophic *Elytrigia repens* – grassland in Viiratsi are significantly higher than those on the border of the *Filipendula-Aegopodium* wet meadow and the alder forest in Porijõgi, being 0.6-7.09 and 0.42-1.05 mg P l^{-1}, respectively. The high P concentrations at Viiratsi are caused by slurry applications to the adjacent field upslope. The high rate of P retention in the Viiratsi site can be explained by: (1) uptake by alders and (2) accumulation in the soil.

In Viiratsi, the total P uptake is about 28% less than that in Porijõgi: 10.8 and 15.1 kg P ha^{-1} yr^{-1} (Tables 3 and 2). In the older stand, half the assimilated P is utilised in leaves; in the younger stand one third. The P retranslocation from senescing leaves in autumn is significantly higher than that for N, being about 60% in both stands. The active soil exchange of P in all buffer zones was negative, *i.e.*, the plant uptake in all communities exceeded the annual accumulation in soil, as for N. However, considering the large soil store of P (1.9-2.0 t ha^{-1} in Porijõgi and 1.4-2.7 t ha^{-1} in Viiratsi; Tables 2 and 3), we suggest that in the long-term most retained P is accumulated in the soil.

In the long-term, this very high loading cannot be balanced by Fe, Al and Ca phosphate precipitation. This is, seemingly, the key process in P retention in Viiratsi. Also, some investigations suggest that permanently high N concentration in soils can cause P leaching (Andrusch *et al.*, 1992). On the other hand, our earlier investigations demonstrate that riparian alder forests are effective buffers for P (Mander *et al.*, 1995). Even in riparian wetlands P can be retained due to micro-scale oxygenation variability within the wetland and, probably, due to phosphorus inactivation by nitrate (see Ripl, 1982).

Our results suggest it is important to harvest older *Alnus incana s*tands (>20 yr) earlier, due to their decreasing uptake and productivity with age.

Removal efficiency of nitrogen and phosphorus in buffer zones

Removal efficiency E (%) of N and P in riparian communities was estimated as:

$$E = 100\% * (Q_{in}C_{in} - Q_{out}C_{out}) / (Q_{in}C_{in}) \qquad (3)$$

where Q_{in} and Q_{out} = inflow and outflow values (m^3 d^{-1}), respectively; C_{in} and C_{out} = concentration values (mg l^{-1}), respectively.

The retention capacity R (kg ha^{-1} yr^{-1}) was calculated as follows:

$$R = \Sigma(Q_{in}C_{in} - Q_{out}C_{out}) / A \qquad (4)$$

where $\Sigma(Q_{in}C_{in} - Q_{out}C_{out})$ is the annual retention and A is the area of the buffer zone.

The specific removal (% m^{-1}) is defined as the removal efficiency per unit width of a buffer zone. This characteristic is useful for planning and establishing buffer communities.

Table 4. Removal efficiency (%), specific removal (% m^{-1}) and retention (kg ha^{-1} yr^{-1}) of nitrogen and phosphorus in test sites.

	Grassland		*Wet meadow*		*Alder forest*		*Whole complex*	
	N	*P*	*N*	*P*	*N*	*P*	*N*	*P*
Porijõgi (less polluted)								
Removal efficiency (%)			36	28	48	67	67	81
Specific removal (% m^{-1})			3.3	2.5	2.4	3.4	2.2	2.6
Retention (kg ha^{-1} yr^{-1})			14.4	0.7	12.4	1.2	13.1	1
Viiratsi (heavily polluted)								
Removal efficiency (%)	70	62	43	46	80	80	97	96
Specific removal (% m^{-1})	6.4	5.6	3.6	3.8	2.9	2.9	1.9	1.9
Retention (kg ha^{-1} yr^{-1})	185.5	6	33.8	1.7	36.2	1.6	56.7	2.2

The buffers investigated showed high removal efficiency and retention values (Table 4). The specific removal of N was decreased downslope which coincides with the edge effect reported in earlier papers (Knauer and Mander, 1989). According to values presented in Table 4, the 50-60 m wide complex buffer zone is able to retain and transform most of nitrogen and phosphorus entering the buffer.

CONCLUSIONS

(1) Riparian wet meadows and grey alder forests *(Alnus incana stands)* are effective buffers on stream banks and lake shores, even with very high input loads (276.3 and 12.8 kg ha^{-1} yr^{-1} of N and P, respectively).

(2) Both N and P retention in more complex sequential buffer zones consisting of different biotopes was higher than in simple sequential buffer zones of fewer biotopes: *e.g.* a 31 m wide buffer zone of wet meadow and grey alder forest removed 67% N and 81% P, within the 51 m buffer zone of grassland strip, wet meadow and alder forest 96% N and 97% P was retained.

(3) Grey alder stands provide potential as wood for fuel. From the point of view of both productivity and nutrient retention, the optimal age to harvest is 12-15 years.

Acknowledgements

This study was supported by the International Science Foundation grants Nos LCU 100 (1994) and LLL100 (1995), and Estonian Science Foundation grant No 2471.

REFERENCES

Andrusch, T., Hupfer, M. and Luther, D. (1992) Chemical and microbial binding forms of phosphorus considering the availability of nitrate in sediment-water systems. Int. Revue ges., *Hydrobiol.*, **77**, 109-120.

APHA (1989) *Standard Methods for the Examination of Water and Waste Water,* 15th edition, American Public Health Organization. 17th edition. Washington.

Balsberg, A.-M. (1982) Plant biomass, primary production and litter disappearance in a *Filipendula ulmaria* meadow ecosystem and the effects of cadmium, *Oikos,* **38**, 72-90.

Bormann, B.T. and Gordon, J.C. (1984) Stand density effects in young red alder plantations: Productivity, photosynthate partitioning and nitrogen fixation. *Ecology,* **65**, 394-402.

Duff, J.H. and Triska, F.J. (1990) Denitrification in sediments from the hyporheic zone adjacent to a small forested stream. *Can. J. Fish. Aquat. Sci.,* **47**, 1140-1147.

Freeze, R.A. and Cherry, J.A. (1979) *Groundwater.* Prentice Hall. 588 p.

Groffmann, P.M. and Tiedje, J.M. (1989) Denitrification in north temperate forest soils: Spatial and temporal patterns at the landscape and seasonal scales. *Soil Biol. Biochem..,* **21**, 613-620.

Groffmann, P.M., Axelrod, E.A., Lemunyon, J.L. and Sullivan, W.M. (1991) Denitrification in grass and forest vegetated filter strips. *J. Environ. Qual.,* **20**, 671-674.

Haycock, N. and Pinay, G. (1993) Nitrate retention in grass and poplar vegetated riparian buffer strips during the winter. *J. Environ. Qual.,* **22**, 273-278.

Huss-Danell, K. and Ohlsson, H. (1992) Distribution of biomass and nitrogen among plant parts and soil nitrogen in a young *Alnus incana* stand. *Can. J. Bot.,* **70**, 1545-1549.

Knauer, N. and Mander, Ü. (1989) Untersuchungen über die Filterwirkung verschiedener Saumbiotope an Gewässern in Schleswig-Holstein. 1. Mitteilung: Filterung von Stickstoff und Phosphor. *Zeitschrift für Kulturtechnik und Landentwicklung,* **30**, 365-376.

Leonardson, L., Bengtsson, L., Davidsson, T., Persson, T. and Emanuelsson, U. (1994) Nitrogen retention in artificially flooded meadows. *Ambio,* **23**, 332-341.

Lowrance, R. (1992) Groundwater nitrate and denitrification in a coastal plain riparian forest. *J. Environ. Qual.,* **21**, 401-405.

Lowrance, R.R., Todd, R., Fail, J., Hendrickson, O., Leonard, R. and Asmussen, L. (l984) Riparian forest as nutrient filters in agricultural watersheds. *BioScience,* **34**, 374-377.

Mander, Ü., Kuusemets, V. and Ivask, M. (1995) Nutrient dynamics of riparian ecotones: A case study from the Porijõgi River catchment, Estonia. *Landscape and Urban Planning,* **31**, 333-348.

Milner, C. and Hughes, R.E. (1968) *Methods for the Measurement of the Primary Production of Grassland.* IBP Handbook No. 6. Blackwell Scientific Publications, Oxford. 70 p.

Peterjohn, W.T. and Correll, D.L. (1984) Nutrient dynamics in an agricultural watershed: observations on the role of a riparian forest. *Ecology,* **65**, 1466-1475.

Pinay, G., Roques, L. and Fabre, A. (1993) Spatial and temporal patterns of denitrification in a riparian forest. *Journal of Applied Ecology,* **30**, 581-591.

Ripl, W. (1982) Nitrification-denitrification and phosphorus inactivation in lake sediments – a reply. *Vatten,* **38**, 363-365.

Rytter, L. (1989) Distribution of roots and root nodules and biomass allocation in young intensively managed grey alder stands on a peat bog. *Plant and Soil,* **119**, 71-79.

Struwe, S. and Kjrller, A. (1990) Seasonality of denitrification in water-logged alder stands. *Plant and Soil,* **128**, 109-113.

Struwe, S. and Kjrller, A. (1991) Denitrification in wet forest soil systems *in situ a*nd in slurry experiments. *For. Ecol. Manage.,* **44** (1), 41-52.

Troelstra, S.R., Wagenaar, R. and Smant, W. (1992) Growth of actinorhizal plants as influenced by the form of nitrogen with special reference to *Myrica gale* and *Alnus glutinosa. J. Exp. Bot.,* **43**, 1349-1359.

Vought, L.B.-M., Dahl, J., Pedersen, C.L. and Lacoursière, J.O. (1994) Nutrient retention in riparian ecotones. *Ambio,* **23**, 342-348.

Weller, D.E., Correll, D.L. and Jordan, T.E. (1994) Denitrification in riparian forests receiving agricultural discharges. In: Mitsch, W.J. (Ed.) *Global Wetlands: Old World and New.* Elsevier Science B.V. p. 117-131.

Yoshinari, T., Hynes, R. and Knowles, R. (1977) Acetylene inhibition of nitrous oxide reduction and measurement of denitrification and nitrogen fixation in soil. *Soil Biol. Biochem.,* **9**, 177-183.

The potential role of riverine wetlands as buffer zones

17

C.A. Johnston[1], J.P. Schubauer-Berigan[2] and S.D. Bridgham[3]

[1] *Natural Resources Research Institute, University of Minnesota, 5013 Miller Trunk Highway, Duluth, MN 55811, USA*
[2] *Belle W. Baruch Institute for Marine Biology and Coastal Research, University of South Carolina, Columbia, SC 29208, USA and Winyah Bay-North Inlet National Research Reserve, P.O. Box 1630, Georgetown, SC 29442, USA*
[3] *Department of Biological Sciences, University of Notre Dame, Notre Dame, IN 46556, USA*

Abstract

Riverine wetlands consist of interacting biological and physical components that substantially alter biogeochemical fluxes. As in other aquatic ecosystems, waterborne material fluxes are a major nutrient source to riverine wetlands, but as in terrestrial ecosystems, biota play a significant role in controlling nutrient retention. Waterborne materials that enter a riverine wetland may be stored, altered by chemical or biological action, or discharged via water or atmospheric fluxes. Sedimentation and denitrification are important processes controlling P and N retention and release by wetlands.

Riverine wetlands are bi-directional buffers: they buffer the effects of surface water runoff on rivers and they buffer the effects of river flooding on adjacent uplands. These effects vary temporally due to seasonal variation in water flow. Water fluxes and fluctuations greatly influence the timing and duration of contact between waterborne nutrients and the ecosystem components that process those nutrients. Riverine wetlands are spatially variable because the dynamic movement of water alters their sediment and vegetation distribution. The spatial variability of riverine wetlands, combined with the spatial and temporal variation of water delivery to them, determines the ability of riverine wetlands to function as metabolic gateways of nutrient and sediment loading to recipient waters.

Wetland position within catchments is one of the most important determinants of stream water quality. Riverine wetlands are favourably positioned to receive non-point source pollutants from upstream urban and agricultural sources. However, this location also places riverine wetlands at risk from urban development, water control structures and excessive pollutant inputs.

INTRODUCTION

Riverine wetlands are fresh to brackish water ecosystems that occur within, and adjacent to, rivers, such that river water is a major hydrologic source for at least a portion of each year. Although riparian forests and some wet grassland meadows occur in similar landscape locations as do riverine wetlands, and may receive river water inputs of short duration during extreme flood events, the flow of surface water in those ecosystems is generally uni-directional from upland to river rather than from river to wetland. Riverine wetlands typically occur in the floodplains of major rivers, but flat terrain riverine wetlands can occur along smaller rivers and streams. This definition of riverine wetland differs from that used in the US wetland classification, in which riverine wetlands are restricted to non-persistent emergent and submergent aquatic beds that occur within river channels (Cowardin et al., 1979).

Some of the world's greatest wetlands are riverine (Table 1). Large wetlands often occur at the mouths of major rivers, typically grading from fresh to brackish water systems at the coast. Extensive wetlands also occur in the floodplains of large rivers, such as the Mississippi and the Amazon, and in large sedimentary basins traversed by rivers, such as The Llanos in Columbia.

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0

Table 1. Major riverine wetland systems of the world
(Sources: Thompson, 1976; Alho *et al.,* 1988; Pinay *et al.,* 1990; Bravard *et al.,* 1992; Vasquez and Wilbert, 1992; Walker, 1992; Dynesius and Nilsson, 1994; Finlayson and Volz, 1994; Healey, 1994; Heliotis *et al.,* 1994; Jensen, 1994; Joly, 1994; Khamphet, 1994; Mitsch *et al.,* 1994; Thaosuwan and Thanakorn, 1994; Xuan, 1994).

Country	*River*	*Type*	*Human Alteration*
Argentina, Brazil, Paraguay, Uruguay	Paraná-Paraguay (The Panatal)	floodplain, delta	dams, deforestation
Australia	Murray-Darling	basin	dams, agriculture
Bangladesh, India	Ganges	delta	agriculture, urbanisation
Botswana	Okavango	basin	deforestation
Brazil	Amazon	floodplain	deforestation
Canada	Frasier	delta	dykes
	Nelson	basin	dams
	St. Lawrence	floodplain	dams
	Peace-Athabasca-Mckenzie	floodplain, delta	
China	Changjiang (Yangtze)	delta	rice paddies, fish ponds
	Zhujiang (Pearl)	delta	rice paddies, fish ponds
	Liaohe	delta	rice paddies, fish ponds
	Huang He (Yellow)	floodplain	agriculture
Colombia, Venezuela	Orinoco (The Llanos)	basin, delta	cattle ranching
Egypt	Nile	floodplain, delta	dams, agriculture
France	Rhone	floodplain, delta	dams, dykes
	Garonne	floodplain	dams, agriculture
Italy	Po	floodplain	dykes
Netherlands, Germany	Rhine	floodplain, delta	polders, agriculture, pollution
Pakistan	Indus	delta	agriculture, urbanisation
Romania, Ukraine	Danube	delta	reed harvesting, agriculture
Russia	Lena	floodplain, delta	
	Ob	floodplain	dams
	Volga	delta	dams, pollution
	Yenisey	floodplain	dams
Spain	Ebro	delta	dams, rice paddies
USA	Atlantic Coastal Plain rivers	floodplain	agriculture
	Gulf Coast rivers	floodplain	agriculture
	Mississippi	floodplain, delta	agriculture, subsidence
	Yukon	floodplain	
Thailand, Laos, Cambodia, Vietnam	Mekong	floodplain, delta	dams, devegetation, agriculture

Riverine wetlands are bi-directional buffers: they buffer the effects of surface water runoff on rivers, and they buffer the effects of river flooding on adjacent uplands. These effects vary temporally due to seasonal variation in water flow, and on shorter timescales due to seiches, tides or storm events. Water fluxes and fluctuations greatly influence the timing and duration of contact between waterborne nutrients and the ecosystem components that process those nutrients, determining a wetland's ability to function as a metabolic gateway of nutrient loading to recipient waters (Wetzel, 1990).

Riverine wetlands are spatially variable because the dynamic movement of water alters their sediment and vegetation distribution. The spatial variability of riverine wetlands, combined with the spatial and temporal variation of water delivery to them, influences their potential for water protection.

RIVERINE WETLAND GEOMORPHOLOGY

The geomorphology of riverine wetlands plays an important role in their maintenance and functioning. Unlike buffer zones in more terrestrial settings, the fluvial forces of erosion and sediment reworking alter the structure of natural riverine wetlands over geologically rapid (hours to centuries) timescales.

River channels

River channels are of three main types: braided, meandering and anastomosing (Salo, 1990). *Braided channels* are characterised by a network of constantly shifting, low sinuosity water courses and are predominantly found in rivers of arid and semi-arid climates, along mountain forelands and along the outwash plains of ice caps and glaciers. Although small wetlands can form in inactive portions of braided channels, the riparian zones adjacent to braided channels are generally not wetlands (NRC 1995). *Meandering channels* occur along low gradient rivers in humid environments and are characterised by a helicoidal flow, a coiling type of water movement that results in erosion of the concave outer bank and deposition along the convex inner bank. Meandering is the result of a river's adjustment to its environment in order to carry its load most efficiently and tends to follow certain mathematical rules with regard to meander wavelength, meander amplitude, stream flow volume, stream flow velocity, channel width, channel depth, floodplain slope and river gradient. Meandering rivers have a single primary channel, high suspended load to bedload ratio, cohesive bank material and relatively steady discharge (Reineck and Singh, 1980). *Anastomosing channels* are characterised by multiple channels that separate and reconnect and occur in large rivers like the Amazon which show a lack of channel competition along their middle and lower reaches (Salo, 1990). This channel form results from a strong flood regime and a dominance of suspended sediments over bedload sediments (Reineck and Singh, 1980).

River floodplains

Floodplains are sedimentary environments, of net deposition, associated with river flooding, created by the processes of stream meandering and overbank flooding (Salo, 1990, Costa *et al.*, 1995). As water flows around a river bend, the current velocity increases on the outer edge of the curve, leading to erosion, and decreases on the inner edge, leading to the deposition of a *point bar.* The lateral and downslope migration of meanders results in the development of *meander scrolls,* in which point bar ridges alternate with low-lying sloughs or swales (Morisawa, 1985). When the migrating river channel cuts the meander from a different angle, a new meander loop may start to form near the old one. As the radius of a new meander loop becomes larger, the old meander may become cut off and abandoned, forming an *oxbow* or *cutoff lake.*

When streams and rivers overflow their banks, sediment is deposited adjacent to the stream channel as a natural levée. Sediment size decreases as current velocity slows with increasing distance from the channel (Salo, 1990). Levées separate the channel from low-lying *backwater a*reas.

Historical maps and aerial photos can be used to document natural channel migration over time (Braga and Gervasoni, 1983, Johnston *et al.*, 1992). This can provide insights into past events that have influenced the present characteristics of riverine wetland soils and vegetation. For example, a map of wetland soils along the East Branch Stream draining into White Clay Lake in northern Wisconsin, USA showed that the location of alluvial soils (Fig. 1a) was anomalous with stream location. However, inspection of historical aerial photos revealed that the contemporary stream outlet had recently migrated to a new location 40 m south of its position for the previous 30 yrs (Fig. 1b), indicating that soil characteristics associated with levée deposition merely lagged behind stream relocation. Historical maps and aerial photos are more commonly used to document artificial channel alteration, such as the channelisation of rivers to prevent meandering (Shäffer, 1973; Sedell and Frogatt, 1984; Fortuné, 1988; Pinay *et al.*, 1990).

Figure 1a. Wetlands bordering the East Branch Stream (second-order) in Cecil, Wisconsin, USA: Soil types described by Johnston *et al.* (1994a). Fluvaquents = alluvial soils, Borosaprists = organic soils, Haplaquents = wet mineral soils. The dashed line shows the contemporary stream channel and the shaded areas are post-glacial beach ridges.

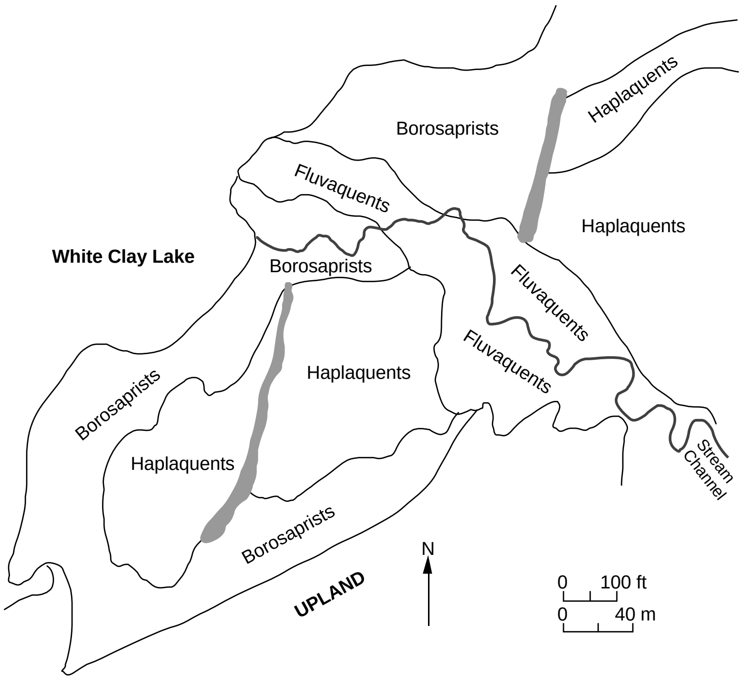
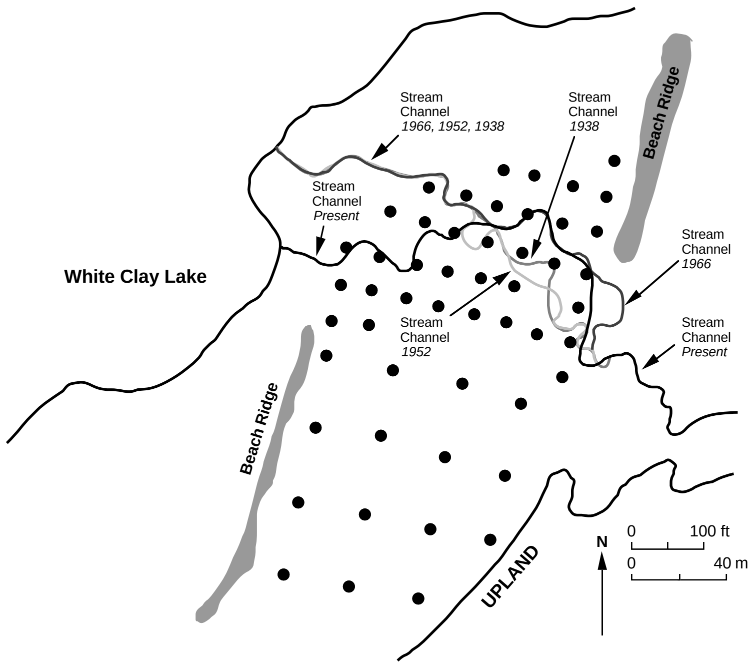


Figure 1b. Wetlands bordering the East Branch Stream (second-order) in Cecil, Wisconsin, USA: Historical stream channel location, determined from aerial photography (after Johnston *et al.,* 1984b).



Some of the largest riverine wetlands occur where rivers flow through flatlands formed by other geomorphic processes. Fluvial forces play a smaller role in the genesis of the river floodplain, but river flooding is the major water source to the wetland. Examples include The Llanos in Columbia, where the Orinoco River flows through a sedimentary basin and the Red-Saskatchewan-Nelson River system in Canada, which flows across former glacial lake beds and the Hudson Bay lowlands. In the eastern US, extensive bottomland hardwood forests and cypress swamps line the rivers that drain the Piedmont and flow across the Coastal Plain to the Atlantic Ocean: the Roanoke, Chowan, Little Pee Dee, Great Pee Dee, Lynches, Black, Santee, Congaree, Altamaha, Cooper, Edisto, Combahee, Coosawhatchie and the Savannah (Mitsch and Gosselink, 1993).

Deltas

Deltas are formed where rivers flow into lakes or oceans, depositing their sediment load as a consequence of decreasing water velocity. Fan-shaped *arcuate deltas,* such as the Nile Delta, have multiple, shifting distributary channels. Branching *birdfoot deltas,* such as the Mississippi delta, form where rivers carrying a large load of suspended sediment flow within a relatively stable distributory, confined by natural levées.

Deltas are associated with some of the world's largest wetlands (Table 1). Deltas contain a combination of salt water, brackish and freshwater wetlands, depending on the relative inputs of ocean and river water. Due to their strategic location at the coastal intersection of major rivers, most deltas are the sites of major cities and have been extensively altered by man. Conversion of the Rhine River Delta to cultivation and urbanisation in The Netherlands, for example, is legendary.

RIVERINE WETLAND SOILS

Sedimentation is an important process in the formation of riverine wetland soils. Measured sedimentation rates range from 0 to 7840 g m^{-2} $year^{-1}$ for various US riverine wetlands (Table 2). Sedimentation rates can be highly variable even within a riverine wetland, as illustrated by a study by Kadlec and Robbins (1984). Within Pentwater Marsh in northern Michigan, USA, mass accumulation was 4700 g m^{-2} $year^{-1}$ on the inside of a channel bend, but only 50 g m^{-2} $year^{-1}$ on the outside of the bend.

Because river floodplains span a range of such sedimentary environments, their soil characteristics occur in a predictable sequence based on current and historical patterns of fluvial geomorphology. Coarse sands and gravels occur where fast-moving channel waters have washed away fine particles, whereas finer sediments are deposited in backwater areas and portions of the floodplain distant from the main channel.

Differences in soil nutrient content accompany variation in soil particle size distribution. Alluvial wetland soils along the East Branch Stream (Fig. 1a) exhibited decreasing phosphorus content as their sand content increased (Table 3). Nitrogen concentrations tended to decrease with increasing sand content, but were also affected by soil microtopography, because nitrification requires aerated soils, and denitrification requires anaerobic soils. Maps of soil nitrate and ammonium within the wetland showed the presence of "hotspots" of very high concentration (Fig. 2). The hotspots for ammonium and nitrate were both located in fine-textured alluvial soils adjacent to the stream, but the NO_3-N hotspot occurred at a location 30 cm higher than the NH_4-N hotspot (Johnston, 1993a). Elevation was positively correlated with NO_3-N ($n = 52$, $r = 0.548$, $P < .01$) and negatively correlated with NH_4-N ($n = 52$, $r = -0.376$, $P < .01$).

Organic matter accumulation also contributes material to soils in quiescent portions of floodplains. The organic matter content of alluvial soils along the East Branch Stream increased as alluvium became finer textured, reflecting the less erosive environment in these locations, and highly organic soils (Histosols) occurred in backwater areas far from the stream (Table 3). Soil organic-N was positively correlated with soil organic matter content ($n = 37$, $r = 0.513$, $P < .01$).

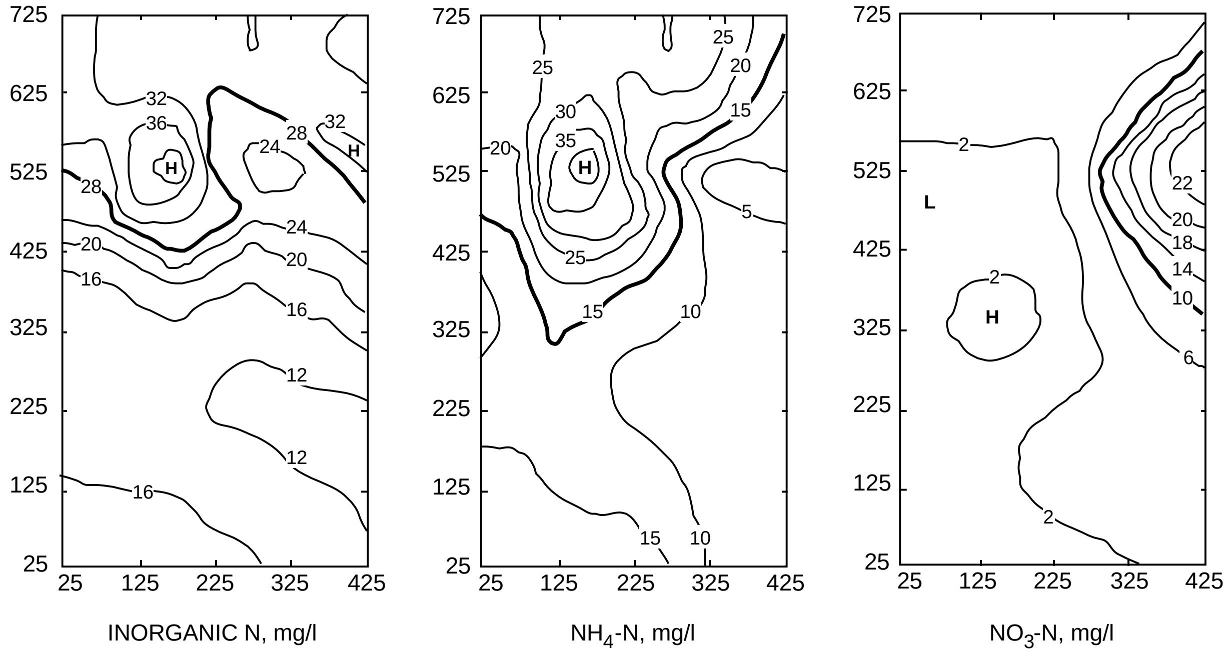
Table 2. Annual thickness and mass accumulation rates for mineral soils in riverine wetlands.

Location	*Wetland description*	*Thickness Accretion cm yr⁻¹*	*Mass Accretion g m⁻² yr⁻¹*	*Method*	*Reference*
Cache River, IL & AR	Alluvial cypress swamp	0.80	5600	Sediment traps	Mitsch *et al*. (1979)
	Levée (3 sites)	0.86	637	Clay pads/sediment disks	Kleiss (1996)
	Swamp (10 sites)	1.55	800	Clay pads/sediment disks	Kleiss (1996)
	Ridge (13 sites)	0.29	174	Clay pads/sediment disks	Kleiss (1996)
	Swale (4 sites)	0.66	221	Clay pads/sediment disks	Kleiss (1996)
Mississippi R., IA	Oxbow lake in floodplain	1.70	-	137 Cs dating	Eckblad *et al*. (1977)
Barataria Basin, LA	Inland with hurricane	0.90	-	Marker horizons	Baumann and Day (1984)
	Streamside without hurricane	1.10	-	Marker horizons	Baumann and Day (1984)
	Streamside with hurricane	1.50	-	Marker horizons	Baumann and Day (1984)
	Barataria Bay marshes	0.60	-	Marker horizons	Baumann and Day (1984)
Fourleague Bay, LA	Inland without hurricane	0.56	-	Marker horizons	Baumann and Day (1984)
	Streamside without hurricane	0.13	-	Marker horizons	Baumann and Day (1984)
Pentwater, MI	River bottom	~0	~0	210 Pb dating	Kadlec and Robbins (1984)
	North channel outside of bend	0.04	50	210 Pb dating	Kadlec and Robbins (1984)
	North channel inside of bend	-	4700	210 Pb dating	Kadlec and Robbins (1984)
	Sedge meadow	0.29	1000	210 Pb dating	Kadlec and Robbins (1984)
	Cattail	0.41	1400	210 Pb dating	Kadlec and Robbins (1984)
	Marsh edge at outlet pool	0.10	110	210 Pb dating	Kadlec and Robbins (1984)
Stevensville, MI	Cranberry Bog inlet stream delta	0.06	330	210 Pb dating	Kadlec and Robbins (1984)
	Cranberry Bog outlet floodplain	0.23	2300	210 Pb dating	Kadlec and Robbins (1984)
Creeping Swamp, NC	Low areas	-	305	Flood event sedimentation	Kuenzler *et al*. (1980)
	Intermediate elevations	-	148	Flood event sedimentation	Kuenzler *et al*. (1980)
	Higher elevations	-	39	Flood event sedimentation	Kuenzler *et al*. (1980)
Cecil, WI	Levée	1.30	7840	137 Cs dating	Johnston *et al*. (1984a)
	Backwater areas	0.50	472	137 Cs dating	Johnston *et al*. (1984a)
Madison, WI	Wet meadow	-	956	Suspended solids budget	Novitzki (1978)

Table 3. Average nutrient and organic matter content of depositional units from five wetland soils (after Johnston, 1984b)

Type of material	*Inorganic P*	*Organic P*	*Total P*	*NH₄-N*	*NO₃-N*	*Organic N*	*Total N*	*Organic Matter*
				mg kg⁻¹				*%*
Sand alluvium	178	42	220	8.4	0.8	1135	1144	2.5
Sandy loam alluvium	325	107	432	6.8	6.5	894	9.7	6.1
Silt loam alluvium	525	401	926	9.8	27.1	7090	7127	12.7
Muck	394	391	786	17.2	4.7	14690	14712	36.7

Figure 2. Maps of soil inorganic nitrogen concentrations in the wetlands bordering the East Branch Stream (after Johnston, 1993). H = concentration dome, L = concentration depression.



RIVERINE WETLAND VEGETATION

The vegetation of riverine wetlands includes life-forms varying from forests to submergent aquatics (Malanson, 1993). Woody plants (trees, shrubs) tend to grow in areas that have shallower water and/or are less frequently flooded, but species such as bald cypress *(Taxodium distichum),* water tupelo *(Nyssa aquatica L.)* and swamp tupelo *(Nyssa sylvatica* var. *biflora)* thrive under inundation (Gill, 1970; Hook and Scholtens, 1978; Whitlow and Harris, 1979; Kozlowski, 1984; Theriot, 1993), and are the dominant trees in south-eastern US riverine forests (McKnight *et al.,* 1981). Tree and shrub genera that commonly grow on river floodplains throughout the world include *Alnus, Fraxinus, Populus, Salix* and *Ulmus* (Gill, 1970).

Emergent and submergent herbaceous plant communities also occur in riverine wetlands. Submergent aquatics usually occur in deep backwater or in areas of the river channel protected from scouring. Emergent aquatics usually occur in water that is less than 80 cm deep. Sedge and grass meadows occur in floodplain areas that are infrequently or shallowly flooded and are used for hay harvesting in some regions of the world.

Riverine wetland vegetation exhibits strong zonation and the mechanisms responsible for this zonation (e.g., competitive ability, environmental gradients, nutrient availability) have been the subject of numerous plant ecological studies. Elevation is related to a number of other environmental variables due to its relationship to water depth and flooding regime and is a frequently cited predictor of riverine wetland plant zonation (Disraeli and Fonda, 1979; Frye and Quinn, 1979; Ewing, 1983; Day *et al.,* 1988; Shipley *et al.,* 1991; Latham *et al.,* 1994). Water velocity and wave action also influences plant community distribution, either directly by seed dispersal and plant mechanical injury, or indirectly through effects on soil erosion, soil texture, and soil organic matter (Nilsson, 1987; Day *et al.*, 1988; Nilsson *et al.,* 1993). Flooding frequency and duration are important determinants of community composition and net primary productivity in riverine wetland forests; intermittent flooding yields higher productivity than does permanent inundation (Bell, 1980; Conner *et al.,* 1981). Soil texture and organic matter content are related to plant zonation within riverine wetlands (Disraeli and Fonda, 1979; Ewing, 1983; Day *et al.,* 1988; Nilsson, 1987; Gaudet and Keddy, 1995) and between wetlands of ascending stream order along longitudinal river gradients (Nilsson *et al.* 1994). Soil and water pH were related to environmental axes in two multivariate statistical studies of riverine wetland plant zonation (Lieffers, 1984; Day *et al.,* 1988), but soil pH was the only one of seven edaphic variables that was <u>not</u> related to plant competitive performance in a greenhouse experiment (Gaudet and Keddy, 1995). Soil fertility influences species distribution in riverine wetlands (Day *et al.*, 1988; Gaudet and Keddy, 1995), whereas salinity may or may not, depending on the range of concentrations present (Disraeli and Fonda, 1979; Ewing, 1983; Lieffers, 1984; Latham *et al.,* 1994). The concentration of pore water sulphide influences NH_4 uptake by *Spartina alternaflora,* a mechanism which could explain variations in its productivity (Bradley and Morris, 1990).

Mats of vegetation often separate from the mineral soil and float on the water surface of riverine wetlands (Lieffers, 1984), an adaptation to the extremes in water level fluctuation associated with many riverine wetlands. Emergent plants that typically form such mats include *Cyperus papyrus, Typha, Phragmites* and *Carex lasiocarpa (*Thompson, 1976; Hogg and Wein, 1988).

Rivers serve as corridors of dispersal for vegetation propagules, a process known as "hydrochory" (Nilsson *et al*., 1993). This efficient transport can be detrimental to endemic wetland flora. However, 235 out of 1100 plant species recorded in riparian habitats of the Adour River, France, were exotics (Pinay *et al*., 1990).

NUTRIENT RETENTION IN RIVERINE WETLANDS

Wetland ecosystems consist of interacting biological and physical components that substantially alter nutrient fluxes. As in other aquatic ecosystems, waterborne material fluxes are a major nutrient source to estuarine and riverine wetlands, but as in terrestrial ecosystems, biota play an important role in nutrient retention. Waterborne materials that enter a wetland may be stored, altered by chemical or biological action, or discharged via water or atmospheric fluxes.

Effects of Hydrology

In streams and rivers, the flow of water greatly influences the timing and duration of contact between waterborne nutrients and the aquatic ecosystems that cycle those nutrients. Rather than cycling nutrients in place, the continuous unidirectional movement of water and materials in streams stretches nutrient cycles into spirals, the length of which is a function of water flow and nutrient exchanges with sediments and biota (Newbold *et al.,* 1981; Elwood *et al.,* 1983). Wetlands within and adjacent to rivers decrease water flow rates and increase nutrient retention times, thereby shortening nutrient spiral lengths (Howard-Williams, 1985).

The flow of water is slower in riverine wetlands than in river channels. Exchanges of nutrients between the water column and sediments are bi-directional in both riverine wetlands and river channels, but water flow may be multi-directional in riverine wetlands, whereas it is generally uni-directional in river channels. The net effect of riverine wetlands on downstream water quality thus depends upon their hydrologic interaction with river water, as well as the nutrient flux and transformation processes that occur within them (Gosselink and Turner, 1978; Novitkzki, 1979; Kadlec *et al.*, 1981; Carter, 1986; Heath, 1992; Groffman, 1994).

Riverine wetland hydrology varies seasonally, especially in areas subject to large intra-annual variation in precipitation or its over-winter storage as snow. Flood amplitudes in unregulated rivers may be as deep as 15 m (Junk, 1993). The high water levels that occur during flood events maximise river water contact with adjacent wetlands. As water levels recede, wetlands that are slightly elevated and distant from the river channel become hydrologically more isolated from the river water. They remain saturated due to stored groundwater and precipitation inputs, but have less contact with materials flowing by in the river, and thus have less of an influence on river water quality. River-wetland water exchanges are minimised during times when surface water in the wetland is frozen.

Riverine wetland hydrology also varies on shorter timescales. Freshwater wetlands on rivers subject to tides and seiches (water surface oscillation in an enclosed basin, such as the Laurentian Great Lakes) experience diurnal water level fluctuations that cause water to flow into and out of them. These diurnal fluctuations are superimposed upon less frequent changes in water level that are driven by storm and snowmelt events. Such water level fluctuations may affect riverine wetland nutrient cycling processes in several ways: (1) by influencing delivery of river water to different parts of the wetland, (2) by influencing fluxes of materials out of the wetland into the river and (3) by establishing alternating aerobic/anaerobic conditions in exposed sediments. Effect #1 could maximise nutrient retention and transformation by increasing the surface area of wetland that is in contact with river water. Effect #3 could maximise conversion of nitrogen from aqueous to gaseous forms by establishing alternating conditions suitable for nitrification and denitrification (Smith and Patrick, 1983), and could also maximise P sediment sorption by formation of amorphous Fe and Al compounds (Sah *et al.*, 1989). However, effect #2 could offset those benefits to water quality by flushing materials back out of the wetland. The net effect of the riverine wetland on river water quality depends on the balance of these influences.

The timing of such water level fluctuations relative to the occurrence of biotic processes within riverine wetlands is also important. If river water flows into the wetland during a time when biota and microbes are dormant, then the potential effects of those organisms on water quality will not be realised. Also, the concentration of soluble nutrients in wetland surface water relative to their concentration in sediments will influence the direction and rate of exchange due to diffusion gradients (Koch and Reddy, 1992).

Although each riverine wetland has a unique hydrology, detailed studies of wetland hydrology in relation to nutrient retention are rare, despite the fact that much generaliseable information can be gained by such studies. Comprehensive studies linking hydrology with riverine wetland nutrient cycling (e.g., Mitsch *et al.* 1979, Kuenzler *et al.* 1980) are among the most cited in the wetland literature. The importance of hydrology as a driving force of wetland functions has been increasingly acknowledged and hydrogeomorphic approaches to wetland classification and management have been adopted by a variety of federal agencies (Brinson, 1993).

Effects of sedimentation

Excess turbidity in surface waters can reduce photosynthesis, decrease oxygen concentrations, impair respiration and feeding of aquatic animals, kill benthic organisms, destroy fish habitat and stimulate the encroachment of exotic and undesirable plant species (Darnell, 1976). When waterborne suspended solids enter a wetland, the decrease in water velocity causes them to settle out onto the soil surface, thereby benefiting downstream water quality. Although vegetation helps to slow the water and filter out particles, mineral sediment deposition is largely a physical settling process.

Mineral sediment deposition has dual benefits to water quality. Not only does it reduce turbidity, but it also retains phosphorus and contaminants that are sorbed to those particles (Stall, 1972). Mineral sediment deposition is a relatively irreversible mechanism; once deposited, the sediments remain part of the soil storage compartment indefinitely.

When mineral sediments or organic matter are deposited in a riverine wetland, associated nutrients and contaminants are also deposited. Based on data from 17 wetland study sites, Johnston (1991) showed that average annual nutrient fluxes via sediment deposition are high (15 g N m^{-2} y^{-1} and 1.5 g P m^{-2} y^{-1}), whereas nutrient fluxes associated with organic soil accumulation are about an order of magnitude lower (1.6 g N m^{-2} y^{-1} and 0.3 g P m^{-2} y^{-1}). The two highest P deposition rates were associated with river floodplain sedimentation (Mitsch *et al.*, 1979, Johnston *et al.*, 1984a). In the study by Mitsch *et al.* (1979), P retention by sediment deposition during the annual spring flood was 3.6 g P m^{-2} y^{-1}, eighteen times more than all other retention mechanisms. However, this amount represented only 4% of the P transported by the river into the wetland, most of which passed through unaltered in river flow.

A two-year study of the riverine Creeping Swamp by Kuenzler *et al.* (1980) illustrates inter-annual variability in P retention and underscores the importance of flood events in wetland P dynamics. Phosphorus retention was twice as high in 1978 (0.73 g P m^{-2} y^{-1}) as in 1977 (0.32 g P m^{-2} y^{-1}), due largely to a flood event that increased sedimentation and P uptake by algae. The wetland retained 61% of the particulate P inputs in 1978, whereas in 1977 particulate P outputs were approximately equal to inputs.

Given their geomorphology, riverine wetlands are well-adapted to sedimentation. Plant species endemic to riverine wetlands survive encrustation and even burial by waterborne silt (Kennedy and Krinard, 1974). However, high sedimentation rates cannot be sustained without geomorphic adjustments, such as the channel migration illustrated in Fig. 1, and excessive sedimentation can cause irreversible wetland degradation.

Nutrient uptake by vegetation

Nutrient concentrations in green tissues generally fall within a narrow range of values, regardless of species (1-3% N and 0.1-0.3% P), so nutrient standing stocks in riverine wetlands are largely a function of plant biomass (Johnston, 1993b). Biomass per unit area varies substantially within and among plant species in wetlands (Johnston, 1988). For example, above-ground biomass of *Typha* measured by a single investigator throughout North America ranged from 0.4-1.3 kg m^{-2}, a 3-fold difference (McNaughton, 1966). An even wider range of values was reported for *Phragmites australis* in Scottish lakes (Ho, 1979). When nutrients are added to wetlands with low natural biomass production, there is typically a change to a more productive species, which allows wetlands to assimilate more nutrients than would be possible by increasing tissue nutrient concentrations alone.

At the end of a growing season, some of the nutrients taken up by plants are retranslocated from green tissues into perennial tissues, some are leached out of the green tissues into wetland surface waters and the rest are returned to the wetland soil surface in litterfall. Thus, the net annual retention of nutrients by vegetation is a fraction of net annual uptake. For example, a study of freshwater riverine marshes in Wisconsin indicated high nutrient uptake by *Scirpus fluviatilis* (20.8 g N m^{-2} y^{-1}, 5.3 g P m^{-2} y^{-1}), but only 26% N and 38% P retention; the rest being returned to the wetland surface in the form of leaching and litterfall (Klopatek, 1978).

Above-ground woody plant tissues generally contain very low nutrient concentrations, averaging only 0.4 %N and 0.01 %P dry weight (Johnston, 1991). However, despite these low concentrations, this storage compartment can be large due to the large amount of woody biomass in wetland forests. In mature forests, woody biomass can contain up to 95 g N m^{-2} and 4.3 g P m^{-2} (Schlesinger, 1978), 10 times the amount of N and 5 times the amount of P normally stored in leaves.

Despite the large nutrient standing stocks in mature trees, net annual retention of nutrients by forests is low in comparison to herbaceous plants (Johnston, 1991). Even though wood can contain large standing stocks of nutrients, the amount of P allocated to wood growth every year is generally less than 0.1 g m^{-2} y^{-1}. In all but one of the studies summarised by Johnston (1991), the majority of nutrients taken up by trees were returned to the wetland surface via leaching and litterfall.

Denitrification

Microbial processes dominate nitrogen transformations and removal in wetlands (Bowden, 1986; Groffman, 1994). Denitrification is of particular interest because it represents a nitrogen removal mechanism with infinite capacity (i.e. nitrogen is completely removed from the wetland without filling an intrasystem pool; Richardson and Nichols, 1985; Richardson, 1989). Denitrification rates have been measured in a wide variety of terrestrial soils and aquatic sediments, but few studies have been conducted in riverine wetlands (Tiedje *et al.*, 1982; Seitzinger, 1988; Johnston, 1991; Seitzinger, 1994). Although highest denitrification rates are usually found in coastal marine and estuarine sediments (2 – 14,938 mg N m^{-2} day^{-1}), rates measured in freshwater sediments and soils are within this range (0.03 – 1428 mg N m^{-2} day^{-1}).

A number of factors are important in regulating rates of denitrification. Of these, temperature and nitrate and carbon availability can directly affect denitrification (Knowles, 1982). Although much is known about how these factors affect the nature and extent of nitrate reduction in particular environments (Payne, 1981; Knowles, 1982; Tiedje *et al.*, 1982), much controversy still exists concerning the primary factors controlling this process spatially within different ecosystems (Robertson *et al.*, 1988; Smith and Duff, 1988; Groffman and Tiedje, 1989; Jorgensen, 1989; Schubauer-Berigan, *et al.* 1991). Tiedje *et al.* (1982) suggest that soil carbon concentrations may actually limit rates of denitrification more than oxygen, since higher carbon concentrations support larger populations of heterotrophs (including denitrifiers) and result in lower soil oxygen concentrations. They further hypothesise that the most important habitats for denitrification are those with an aerobic-anaerobic interface, particularly saturated soils or aquatic sediments. When carbon is not limiting, denitrification is limited by the rate of diffusion of nitrate from active zones of nitrification (aerobic zones) to active zones of denitrification (anaerobic zones) (Reddy *et al.*, 1976). In sediments, the source of nitrate can be *in situ* microbial nitrification (which can be limited by the supply of ammonia) or nitrate from the water column in contact with sediments. Other factors, such as nitrogen binding patterns in sediments (Seitzinger *et al.*, 1991), and methane (Roy *et al.*, 1994) and sulphide inhibition of nitrifiers (Joyce and Hollibaugh, 1995), can also affect the supply of nitrate available for denitrification. These factors also play a role in the coupling and fluxes of nitrogen between the water column and sediments in wetlands. As pointed out by Groffman (1994), although wetlands are often assumed to have high rates of denitrification, N and C may actually be too poor to support microbial denitrification.

We measured denitrification potentials from 158 soil samples collected in spring and summer from two riverine wetlands of the St. Louis River system, which borders Minnesota and Wisconsin, USA (Schubauer-Berigan *et al.*, 1995). We were initially surprised by the low denitrification potentials, even in the summer. However, these results corresponded well with low pore-water nitrate concentrations measured during these periods, suggesting that denitrification in these wetlands is nitrate limited. Nitrate additions caused large increases in denitrification rates.

ALTERATION AND RESTORATION OF RIVERINE WETLANDS

Humans have extensively manipulated the world's rivers, often destroying and degrading the associated wetlands. Seventy-seven percent of the total water discharge of the 139 largest river systems in the northern third of the world is moderately to strongly affected by dams and by water regulation resulting from reservoir operation, inter-basin diversion and irrigation (Dynesius and Nilsson, 1994). Humans have also drained vast riverine wetlands for agricultural production. In the US, examples of riverine wetlands converted to agriculture include: the Great Kankakee Marsh, which formerly occupied 13,700 km^2 of Illinois and Indiana (Mitsch and Gosselink, 1993); 6.5 million ha of bottomland hardwood forests of the lower Mississippi River, converted primarily to soybean production (Nature Conservancy, 1992) and extensive areas of bottomland hardwoods in the Southeast Coastal Plain. As a consequence of such alterations, riverine wetlands, especially those with unregulated hydrology, are rare. This has resulted in an unknown, but probably significant, loss of their associated water-quality functions.

Given the increasing dependence of humans on rivers for water supply (Postel *et al.*, 1996) and waste disposal (Smith *et al.*, 1987; Cole *et al.*, 1993), it is certain that riverine wetland buffer zones will continue to be destroyed and degraded. Wetlands in headwater areas, in oxbows and in low-velocity channels have not been extensively restored, despite recognition of the ecological and hydrological importance of the connection between rivers and their floodplains (NRC, 1992). River fluctuation is crucial to the maintenance of riverine wetlands, hence their restoration requires removal of dams, channelisation and other massive public works. There are notable exceptions, such as the Des Plaines River wetlands demonstration project (Hey *et al.*, 1989), the Illinois River-Floodplain (Sparks, 1992), the Kissimmee Riverine-Floodplain (Berger, 1992), and small restoration projects in the lower Mississippi River floodplain (Sharitz, 1992), but these are the exceptions rather than the rule.

REFERENCES

Alho, C.J.R., Lacher Jr. T.E. and Gonçalves, H.C. (1988) Environmental degradation in the Panatal ecosystem. *BioScience*, **38**(3), 164-171.

Baumann, R.H. and Day, Jr. J.W. (1984) Mississippi deltaic wetland survival: sedimentation versus coastal subsidence. *Science*, **224**, 1093-1095.

Bell, D.T. (1980) Gradient trends in the streamside forests of central Illinois. *Bulletin of the Torrey Botanical Club*, **107**, 172-180.

Berger, J.J. (1992) The Kissimmee Riverine-Floodplain system. In: National Research Council (NRC). *Restoration of Aquatic Ecosystems.* National Academy Press, Washington, DC. pp. 477-496.

Bowden, W.B. (1986) Nitrification, nitrate reduction and nitrogen immobilization in a tidal freshwater marsh sediment. *Ecology*, **67**(1), 88-99.

Bradley, P.M. and J.T. Morris. (1990) Influence of oxygen and sulfide concentration on nitrogen kinetics in Spartina alternaflora. *Ecology*, **71**(1), 282-287.

Braga, G. and Gervasoni, S. (1983) Evoluzióne stòrica dell'àlveo del fiume Po nel territòrio Lodigiano. Piacentino: rischi iderogeòlogici connessi. Atti Convegno Naz. Il suòlo come risórsa **60**, 59-69.

Bravard, J.P., Roux, A.L., Amoros, C. and Reygrobellet, J.L. (1992) The Rhône River: a large alluvial temperate river. In: Calow, P. and Petts, G.E. (Eds.) *The Rivers Handbook: Hydrological and Ecological Principles.* Blackwell, Oxford. pp. 426-447.

Brinson, M.M. (1993) *A hydrogeomorphic classification for wetlands.* Water Research Program Technical Report WRP-DE-4. U.S. Army Corps of Engineers Waterway Experiment Station, Vicksburg, MS.

Carter, V. (1986) An overview of the hydrologic concerns related to wetlands in the United States. *Can. J. Bot.*, **64**, 364-374.

Cole, J.J., Peierls, B.L., Caraco, N.F. and Pace, M.L. (1993) Nitrogen loading of rivers as a human-driven process. In: McDonnell, M.J. and Pickett, S.T.A. (Eds.) *Humans as components of ecosystems: the ecology of subtle human effects and populated areas.* Springer-Verlag, New York. pp. 141-157.

Conner, W.H., Gosselink, J.G. and Parrondo, R.T. (1981) Comparison of the vegetation of three Louisiana swamp sites with different flooding regimes. *Amer. J. Bot.*, 320-331.

Costa, J.E., Miller, A.J., Potter, K.W. and Wilcock, P.R. (1995) Natural and anthropogenic influences in fluvial geomorphology. *Geophysical Monogr.*, **89**, American Geophysical Union, Washington, DC.

Cowardin, L.M., Carter, V., Golet, F.C. and LaRoe, E.T. (1979) *Classification of wetlands and deepwater habitats of the United States.* U.S. Fish and Wildlife Service, FWS/OBS-79/31. pp. 103.

Day, R.T., Keddy, P.A., McNeill, J. and Carleton, T. (1988) Fertility and disturbance gradients: a summary model for riverine marsh vegetation. *Ecology*, **69**, 1044-1054.

Darnell, R.M. (1976) *Impacts of construction activities in wetlands of the United States.* Ecological Research Series EPA-600/3-76-045, U.S. Environmental Protection Agency, Corvallis, OR.

Disraeli, D.J. and Fonda, R.W. (1979) Gradient analysis of the vegetation in a brackish marsh in Bellingham Bay, Washington. *Can. J. Bot.*, **57**, 465-475.

Dynesius, M. and Nilsson, C. (1994) Fragmentation and flow regulation of river systems in the northern third of the world. *Science*, **266**, 753-762.

Eckblad, J.W., Peterson, N.L., Ostlie, K. and Temte, A. (1977) The morphometry, benthos and sedimentation rates of a floodplain lake in Pool 9 of the upper Mississippi River. *Am. Midl. Nat.*, **97**, 433-443.

Elwood, J.W., Newbold, J.D., O'Neill, R.V. and van Winkle, W. (1983) Resource spiralling: an operational paradigm for analysing lotic ecosystems. In: Fontaine, T.D. and Bartell, S.M. (Eds.) *Dynamics of Lotic Ecosystems.* Ann Arbor Science Publishers, Ann Arbor, MI. pp. 3-27.

Ewing, K. (1983) Environmental controls in Pacific Northwest intertidal marsh plant communities. *Can. J. Bot.*, **61**, 1105-1116.

Finlayson, C.M. and Volz, J. (1994) A strategic action plan for the conservation of wetlands of the Lower Volga, Russia. In Mitsch, W.J. (Ed.) *Global Wetlands: Old World and New.* Elsevier Science, Amsterdam. pp. 729-736.

Fortuné, M. (1988) Historical changes of a large river in an urban area: the Garonne River, Toulouse, France. *Regulated Rivers,* **2**, 179-186.

Frye, R.J. and Quinn, J.A. (1979) Forest development in relation to topography and soils in a floodplain of the Raritan River, New Jersey. *Bulletin of the Torrey Botanical Club,* **106**, 334-345.

Gaudet, C.L. and Keddy, P.A. (1995) Competitive performance and species distribution in shoreline plant communities: a comparative approach. *Ecology,* **76**, 280-291.

Gill, C.J. (1970) The flooding tolerance of woody species – a review. *Forestry Abstracts,* **31**, 671-688.

Gosselink, J.G. and Turner, R.E. (1978) The role of hydrology in freshwater wetland ecosystems. In: *Freshwater Wetlands: Ecological Processes and Management Potential.* Academic Press, New York. pp. 63-78.

Groffman, P.M. (1994) Denitrification in freshwater wetlands. In: *Current Topics in Wetland Biogeochemistry,* Vol. 1. Wetland Biogeochemistry Institute, Louisiana State University, Baton Rouge. pp. 15-35.

Groffman, P.M. and Tiedje, J.M. (1989) Denitrification in North Temperate forest soils: relationships between denitrification and environmental factors at the landscape scale. *Soil Biol. Biochem.,* **21**, 621-626.

Healey, M.C. (1994) Effects of dams and dikes on fish habitat in two Canadian river deltas. In: Mitsch, W.J. (Ed.) *Global Wetlands: Old World and New.* Elsevier Science, Amsterdam. pp. 385-398.

Heath, R.T. (1992) Nutrient dynamics in Great Lake coastal wetlands: future directions. *J. Great Lakes Res.,* **18**(4), 590-602.

Heliotis, F.D., Vellidis, G., Bandacu, D. and Pringle, C.M. (1994) The Danube Delta: historical wetland drainage and potential for restoration. In Mitsch, W.J. (Ed.) *Global Wetlands: Old World and New.* Elsevier Science, Amsterdam. pp. 759-767.

Hey, D.L., Cardamone, M.A., Sather, J.H. and Mitsch, W.J. (1989) Restoration of riverine wetlands: the Des Plaines River wetlands demonstration project. In: Mitsch, W.J. and Jørgensen, S.E. (Eds.) *Ecological Engineering: An Introduction to Ecotechnology.* John Wiley & Sons, New York. pp. 159-163.

Ho, Y.B. (1979) Short development and production studies of Phragmites australis (Cav.) Trin. ex Steudel in Scottish lochs. *Hydrobiologia,* **64**, 215-222.

Hogg, E.H. and Wein, R.W. (1988) The contribution of Typha components to floating mat buoyancy. *Ecol.,* **69**, 1025-1031.

Hook, D.D. and Scholtens, J.R. (1978) Adaptations and flood tolerance of tree species. In: Hook, D.D. and Crawford, R.M.M. (Eds.) *Plant Life in Anaerobic Environments.* Ann Arbor Sci. Publ., Ann Arbor, MI. pp. 299-331.

Howard-Williams, C. (1985) Cycling and retention of nitrogen and phosphorus in wetlands: a theoretical and applied perspective. *Freshwater Biology,* **15**, 391-431.

Jensen, A.E. (1994) Ecotones project sites in the Murray-Darling Basin, Australia. In: Jensen, A. (Ed.) *Ecotones at the River Basin Scale – Global Land/Water Interactions.* South Australian Department of Environment and Natural Resources, in association with UNESCO Ecotones Research Project 'Role of Land/Inland Water Ecotones in Landscape Management and Restoration', Adelaide, Australia. pp. 118-146.

Johnston, C.A. (1988) *Productivity of wet soils: Biomass of cultivated and natural vegetation.* Oak Ridge National Laboratory Report #ORNL/Sub/84-18435/1.

Johnston, C.A. (1991) Sediment and nutrient retention by freshwater wetlands: effects on surface water quality. *Critical Reviews in Environmental Control,* **21**, 491-565.

Johnston, C.A. (1993a) Material fluxes across wetland ecotones in northern landscapes. *Ecological Applications,* **3**, 424-440.

Johnston, C.A. (1993b) Mechanisms of wetland-water quality interaction. In: Moshiri, G.A. (Ed.) *Constructed Wetlands for Water Quality Improvement.* Lewis Press, Chelsea, MI. pp. 293-299.

Johnston, C.A., Bubenzer, G.D., Lee, G.B., Madison, F.W. and McHenry, R.J. (1984a) *Nutrient trapping by sediment deposition in a seasonally flooded lakeside wetland. J. Environ. Qual.,* **13**, 283-290.

Johnston, C.A., Lee, G.B. and Madison, F.W. (1984b) The stratigraphy and composition of a lakeside wetland. *Soil Sci. Soc. Amer. J.,* **48**, 347-345.

Johnston, C.A., Pastor, J. and Pinay, G. (1992) Quantitative methods for studying landscape boundaries. In: di Castri, F. and Hansen, A. (Eds.) *Landscape Boundaries: Consequences for Biotic Diversity and Ecological Flows.* Springer-Verlag, New York. pp. 107-125.

Joly, C.A. (1994) Biodiversity of the gallery forest and its role in soil stability in the Jacaré-Pepira water, state of Sao Paulo, Brazil. In: Jensen, A. (Ed.) *Ecotones at the River Basin Scale – Global Land/Water Interactions.* South Australian Department of Environment and Natural Resources, in association with UNESCO Ecotones Research Project 'Role of Land/Inland Water Ecotones in Landscape Management and Restoration', Adelaide, Australia. pp. 40-66.

Jorgensen, K.S. (1989) Annual pattern of denitrification and nitrate ammonification in estuarine sediment. *Appl. Environ. Microbiol.,* **55**, 1841-1847.

Joyce, S.B. and Hollibaugh, J.T. (1995) Influence of sulfide inhibition of nitrification on nitrogen regeneration in sediments. *Science,* **270**, 623-625.

Junk, W.J. (1993) Wetlands of tropical South America. In: Whigham, D.F. *et al.* (Eds.) *Wetlands of the World.* Kluwer, Dordrecht, The Netherlands. pp. 679-739.

Kadlec, R.H. and Robbins, J.A. (1984) Sedimentation and sediment accretion in Michigan coastal wetlands (U.S.A.). *Chem. Geol.*, **44**, 119-150.

Kadlec, R.H., Hammer, D.E., Nam, I-S. and Wilkes, J.O. (1981) The hydrology of overland flow in wetlands. *Chem. Eng. Commun.*, **9**, 331-344.

Kennedy, H.E. and Krinard, J.S. (1974) *1973 Mississippi River Flood's Impact on Natural Hardwood Forests and Plantations.* U.S. Forest Service Research Note 5O-177.

Khamphet, R. (1994) Ecotone studies carried out in selected wetlands in Lao PDR. In: Jensen, A. (Ed.) *Ecotones at the River Basin Scale – Global Land/Water Interactions.* South Australian Department of Environment and Natural Resources, in association with UNESCO Ecotones Research Project 'Role of Land/Inland Water Ecotones in Landscape Management and Restoration', Adelaide, Australia. pp. 180-186.

Kleiss, B.A. (1996) Sediment retention in a bottomland hardwood wetland in eastern Arkansas. *Wetlands,* **16**, 321-333.

Klopatek, J.M. (1978) Nutrient dynamics of freshwater riverine marshes and the role of emergent macrophytes. In: Good, R.E., Whigham, D.F. and Simpson, R.L. (Eds.) *Freshwater Wetlands: Ecological Processes and Management Potential.* Academic Press, New York. pp. 195-216.

Knowles, R. (1982) Denitrification. *Microbiol. Revs.*, **46**, 43-70.

Koch, M.S. and Reddy, K.R. (1992) Distribution of soil and plant nutrients along a trophic gradient in the Florida Everglades. *Soil Sci. Soc. Am. J.*, **56**, 1492-1499.

Kozlowski, T.T. (1984) Responses of woody plants to flooding. In: *Flooding and Plant Growth.* Academic Press, New York. pp. 129-163.

Kuenzler, E.J., Mulholland, P.J., Yarbro, L.A. and Smock, L.A. (1980) *Distributions and budgets of carbon, phosphorus, iron and manganese in a floodplain swamp ecosystem.* Water Resources Research Institute Report No. 157, University of North Carolina, Chapel Hill.

Latham, P.J., Pearlstine, L.G. and Kitchens, W.M. (1994) Species association changes across a gradient of freshwater, oligohaline and mesohaline tidal marshes along the lower Savannah River. *Wetlands,* **14**, 174-183.

Lieffers, V.J. (1984) Emergent plant communities of oxbow lakes in northeastern alberta: salinity, water-level fluctuation and succession. *Can. J. Bot.*, **62**, 310-316.

Malanson, G.P. (1993) *Riparian Landscapes.* Cambridge University Press, Cambridge.

McKnight, J.S., Hook, D.D., Langdon, O.G. and Johnson, R.L. (1981) Flood tolerance and related characteristics of trees of the bottomland forests of the southern United States. In: Clark, J.R. and Benforado, J. (Eds.) *Wetlands of Bottomland Hardwood Forests.* Elsevier, Amsterdam. pp. 29-44.

McNaughton, S.J. (1966) Ecotype function in the Typha community type. *Ecol. Monogr.*, **36**, 297-325.

Mitsch, W.J. and Gosselink, J.G. (1993) *Wetlands* (second edition). Van Nostrand Reinhold, New York.

Mitsch, W.J., Dorge, C.L. and Wiemhoff, J.R. (1979) Ecosystem dynamics and a phosphorus budget of an alluvial cypress swamp in southern Illinois. *Ecology,* **60**, 1116-1124.

Mitsch, W.J., Mitsch, R.H. and Turner, R.E. (1994) Wetlands of the Old and New Worlds: ecology and management. In: Mitsch, W.J. (Ed.) *Global Wetlands: Old World and New.* Elsevier Science, Amsterdam. pp. 3-56.

Morisawa, M. (1985) *Rivers: Form and Process.* Geomorphology Texts 7. Longman Group, Hong Kong.

National Research Council (NRC) (1992) *Restoration of Aquatic Ecosystems.* National Academy Press, Washington, DC.

National Research Council (NRC) (1995) *Wetlands: Characteristics and Boundaries.* National Academy Press, Washington, DC.

Nature Conservancy (1992) *The Forested Wetlands of the Mississippi River: An Ecosystem in Crisis.* The Nature Conservancy, Baton Rouge, Louisiana.

Newbold, J.D, Elwood, J.W., O'Neill, R.V. and van Winkle, W. (1981) Measuring nutrient spiralling in streams. *Can. J. Fish. Aquat. Sci.*, **38**, 860-863.

Nilsson, C. (1987) Distribution of stream-edge vegetation along a gradient of current velocity. *J. Ecol.,* **75**, 513-522.

Nilsson, C., Ekblad, A., Dynesius, M., Backe, S., Gardfjell, M., Carlberg, B., Hellquist, S. and Jansson, R. (1994) A comparison of species richness and traits of riparian plants between a major river channel and its tributaries. *J. Ecol.,* **82**, 281-295.

Nilsson, C., Nilsson, E., Johansson, M.E., Dynesius, M., Grelsson, G., Xiong, S., Jansson, R. and Danvind, M. (1993) Processes structuring riparian vegetation. *Current Topics in Botanical Research*, **1**, 419-431.

Novitzki, R.P. (1978) *Hydrology of the Nevin wetland near Madison, Wisconsin.* U.S. Geological Survey, Water Resources Investigation 78-48, U.S. Department of Commerce PB-284 118.

Novitzki, R.P. (1979) Hydrologic characteristics of Wisconsin's wetlands and their influence on floods, stream flow and sediment. In: Greeson, P.E., Clark, J.R. and Clark, J.E. (Eds.) *Wetland Functions and Values: The State of Our Understanding.* American Water Resources Assoc., Minneapolis, MN. pp. 377-388.

Payne, W.J. (1981) *Denitrification.* John Wiley and Sons, New York, N.Y.

Pinay, G., Décamps, H., Chauvet, E. and Fustec, E. (1990) Functions of ecotones in fluvial systems. In: Naiman, R.J. and Décamps, H. (Eds.) *The Ecology and Management of Aquatic-Terrestrial Ecotones.* Man and the Biosphere Series, Vol. 4. UNESCO, Paris, France. pp. 141-169.

Postel, S.L., Daily, G.C. and Erlich, P.R. (1996) Human appropriation of renewable fresh water. *Science*, **271**, 785-788.

Reddy, K.R., Patrick Jr., W.H. and Phillips, R.E. (1976) Ammonium diffusion as a factor in nitrogen loss from flooded soils. *Soil Sci. Soc. Am. J.*, **40**, 528-533.

Reineck, H.E. and Singh, I.B. (1980) *Depositional sedimentary environments with reference to terrigenous clastics.* Second Edition. Springer-Verlag. Berlin and Heidelberg, Germany.

Richardson, C.J. (1989) Freshwater wetlands: transformers, filters, or sinks? In Sharitz, R.R. and Gibbons, J.W. (Eds.) *Freshwater Wetlands and Wildlife,* CONF-8603101, DOE Symposium Series No. 61, USDOE Office of Scientific and Technical Information, Oak Ridge, TN. pp. 25-46.

Richardson, C.J. and Nichols, D.S. (1985) Ecological analysis of wastewater management criteria in wetland ecosystems. In: Godfry, P.J., Kaynor, E.R., Pelczarski, S. and Benforado, J. (Eds.) *Ecological Considerations in Wetlands Treatment of Municipal Wastewaters.* Van Nostrand Reinhold, New York. pp. 351-391.

Robertson, G.P., Huston, M.A., Evans F.C. and Tiedje, J.M. (1988) Spatial variability in a successional plant community: patterns of nitrogen availability. *Ecology,* **69**, 1517-1524.

Roy, R., Legendre, P., Knowles, R. and Charlton, M.N. (1994) Denitrification and methane production in sediment of Hamilton Harbour (Canada). *Microb. Ecol.*, **27**, 123-141.

Sah, R.N., Mikkelsen D.S. and Hafez, A.A. (1989) Phosphorus behavior in flooded-drained soils. II. Iron transformation and phosphorus sorption. *Soil Science Society of America Journal,* **53**, 1723-1729.

Salo, J. (1990) External processes influencing origin and maintenance of inland water-land ecotones. In: Naiman, R.J. and Décamps, H. (Eds.) *The Ecology and Management of Aquatic-Terrestrial Ecotones.* Man and the Biosphere Series, Vol. 4. UNESCO, Paris, France. pp. 37-64.

Schlesinger, W.H. (1978) Community structure, dynamics and nutrient cycling in the Okefenokee cypress swamp-forest. *Ecol. Monogr.*, **48**, 43-65.

Schubauer-Berigan, J.P., Capone, D.G., Cockran, K., Kazumi, J. and Epler, N. (1991) *Bacterial transformations of nitrate and aldicarb (TEMIC) in groundwaters of Long Island.* Final Report submitted to U.S.G.S.

Schubauer-Berigan, J.P., Johnston, C.A. and Bridgham, S.D. (1995) *Physical, chemical and biological processes in riverine wetlands* (abstract). 87th Annual Meeting, American Society of Agronomy. St. Louis, MO.

Sedell, J.R. and Froggatt, J.L. (1984) Importance of streamside forests to large rivers: the isolation of the Willamette River, Oregon, USA, from its floodplain by snagging and streamside forest removal. *Verhandlungen Internationale Vereinigung Limnologie,* **22**, 1828-1834.

Seitzinger, S.P. (1988) Denitrification in freshwater and coastal marine ecosystems: Ecological and geochemical significance. *Limnol. Oceanogr.*, **33**, 702-724.

Seitzinger, S.P. (1994) Linkages between organic matter mineralization and denitrification in eight riparian wetlands. *Biogeochemistry,* **25**, 19-39.

Seitzinger, S.P., Gardner W.S. and Spratt, A.K. (1991) The effect of salinity on ammonium sorption in aquatic sediments: implications for benthic nutrient recycling. *Estuaries,* **14**, 167-174.

Shäffer, W. (1973) Altrhein verbund am nördlichen Oberrhein. *Courier Forschungsinstitut Senckenberg,* **7**, 1-63.

Sharitz, R. (1992) Bottomland hardwood wetland restoration in the Mississippi drainage. In: National Research Council (NRC). *Restoration of Aquatic Ecosystems.* National Academy Press, Washington, DC. pp. 496-505.

Shipley, B., Keddy P.A. and Lefkovitch, L.P. (1991) Mechanisms producing plant zonation along a water depth gradient: a comparison with the exposure gradient. *Can J. Bot.*, **69**, 1420-1424.

Smith, C.J. and Patrick Jr., W.H. (1983) Nitrous oxide emission as affected by alternate anaerobic and aerobic conditions from soil suspensions enriched with ammonium sulfate. *Soil Biol. Biochem.*, **15**(6), 693-697.

Smith, R.A., Alexander R.B. and Wolman, M.G. (1987) Water-quality trends in the nation's rivers. *Science,* **235**, 1607-1615.

Smith, R.L. and Duff, J.H. (1988) Denitrification in a sand and gravel aquifer. *Appl. Environ. Microbiol.*, **54**, 1071-1078.

Sparks, R. (1992) The Illinois River-Floodplain ecosystem. In: National Research Council (NRC). *Restoration of Aquatic Ecosystems.* National Academy Press, Washington, DC. pp. 412-432.

Stall, J.B. (1972) Effects of sediment on water quality. *J. Environ. Quality,* **1**, 353-360.

Thaosuwan, A. and Thanakorn, U. (1994) Wetland management in Thailand. In: Jensen, A. (Ed.) *Ecotones at the River Basin Scale – Global Land/Water Interactions.* South Australian Department of Environment and Natural Resources, in association with UNESCO Ecotones Research Project 'Role of Land / Inland Water Ecotones in Landscape Management and Restoration', Adelaide, Australia. pp. 187-205.

Theriot, R.F. (1993) *Flood tolerance of plant species in bottomland forests of the Southeastern United States.* Wetlands Research Program Technical Report WRP-DE-6. U.S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg, MS.

Thompson, K. (1976) Swamp development in the head waters of the White Nile. In: Rzóska, J. (Ed.) *The Nile, Biology of an Ancient River.* Dr. W. Junk, The Hague, The Netherlands. pp. 177-196.

Tiedje, J.M., Sexstone, A.J., Myrold D.D. and Robinson, J.A. (1982) Denitrification: ecological niches, competition and survival. *Antonie van Leeuwenhoek,* **48**, 569-583.

Vasquez, E. and Wilbert, W. (1992) The Orinoco: physical, biological and cultural diversity of a major tropical alluvial river. In: Calow, P. and Petts, G.E. (Eds.) *The Rivers Handbook: Hydrological and Ecological Principles.* Blackwell, Oxford. pp. 448-471.

Walker, K.F. (1992) The River Murray, Australia: a semiarid lowland river. In: Calow, P. and Petts, G.E. (Eds.) *The Rivers Handbook: Hydrological and Ecological Principles.* Blackwell, Oxford. pp. 472-492.

Wetzel, R.G. (1990) Land-water interfaces: Metabolic and limnological regulators. *Verh. Internat. Verein. Limnol.*, **24**, 6-24.

Whitlow, T.H. and Harris, R.W. (1979) *Flood tolerance in plants: a state-of-the-art review.* Technical Report E-79-2. U.S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg, MS.

Xuan, T.T. (1994) The preliminary results of a study of wetlands in the Mekong Delta, Vietnam. In: Jensen, A. (Ed.) *Ecotones at the River Basin Scale – Global Land/Water Interactions.* South Australian Department of Environment and Natural Resources, in association with UNESCO Ecotones Research Project 'Role of Land/Inland Water Ecotones in Landscape Management and Restoration', Adelaide, Australia. pp. 167-179.

The Potential Role of Mid–Field Forests as Buffer Zones

18

L. Ryszkowski, A. Bartoszewicz and A. Kedziora

Research Centre for Agricultural and Forest Environments, Polish Academy of Science, ul. Bukowska 19, 60809 Poznan, Poland

Abstract

A method for estimating evapotranspiration in forested areas with climatological data was used to delimit the width of buffer zones, known as the biogeochemical barrier (BGCB), within which the influx of mineral nitrogen from the field is removed. The BGCB width in pine forest is 6 m and in birch forest is 11 m. In the BGCB of birch forest (1 m-11 m) mineral nitrogen retention of 211 g y^{-1} was observed and in pine forest the retention was 274 g y^{-1}. It is argued that within the BGCB the total input of mineral nitrogen from the field is removed. Nitrogen accumulation in soil organic matter was estimated to be 1.4 g m^{-2} y^{-1} in pine forest and 1.1 g m^{-2} y^{-1} in birch forest. Long-term storage in bole wood is about 1% of the withdrawal in the soil organic matter. About 94-97% of nitrogen removed from subsurface water flow in the buffer zone is incorporated in the dynamic internal nitrogen cycle of retranslocation, release in decomposition and re-use by root systems as well as in cation exchange (NH_4^+). It is also released in denitrification or NH_3 volatilisation. The hypothesis is put forward that plant uptake is responsible for the removal of mineral nitrogen from the groundwater, while denitrification is responsible for the release of nitrogen from forested areas to the atmosphere.

Studies on the influence of the plant structure of the watershed area on nitrate concentrations in outlets from drainage ditches provided empirical equations for forecasting nitrate concentration in various catchments located on Hapludalf and Udipsammens soils, provided that the buffer zone area in each watershed is known.

INTRODUCTION

Several studies have shown retention of elements by vegetated riparian buffer strips (e.g. Pauliukevicius, 1981; Lowrance *et al.*, 1983; Peterjohn and Correll, 1984; Pinay and Decamps, 1988; Knauer and Mander, 1989; Ryszkowski *et al.*, 1989; Cooper, 1990; Muscutt *et al.*, 1993; Hillbricht-Ilkowska *et al.*, 1995). Despite some findings that riparian strips could sometimes be a source of elements to draining waters (Lee *et al.*, 1975; Bartoszewicz, 1990; 1994; Cooper, 1990; Vanek, 1991), there is no doubt that, in general, riparian buffer strips control the output of chemical pollutants from a catchment area (Lowrance *et al.*, 1985; Baker, 1992; Muscutt *et al.*, 1993). There is also an increasing body of information to show that shelterbelts (rows of trees) and mid-field forests, located in upland parts of a catchment area, can influence subsurface water chemistry by removing elements from the groundwater that is in reach of their roots (Lowrance *et al.*, 1983; Ryszkowski and Bartoszewicz, 1989; Bartoszewicz, 1990; Ryszkowski, 1994).

The majority of studies on buffer zones rely on input-output analyses and do not attempt to study in detail the internal processes responsible for any observed effects. Much more information has been gathered on the control of surface flows by riparian buffer strips than of subsurface discharges (see e.g. Baker, 1992; Muscutt *et al.*, 1993; Hillbricht-Ilkowska *et al.*, 1995). Retention mechanisms of surface flow include precipitation of insoluble particles, sorption of metal ions, uptake by strip vegetation and, in relation to nitrogen, volatilisation (evolution of ammonia or N_2 and NO_x), especially when the surface is flooded. Mechanisms controlling the passage of chemical compounds in subsurface water are less understood, but there is a general assumption that the following processes are important: ion exchange, plant uptake and, in relation to nitrogen, denitrification provided that there are substantial

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0 © 1997 Quest Environmental.

inputs of readily mineralisable nitrate as well as energy-rich organic compounds and provided that anoxic conditions prevail (Lowrance *et al.,* 1985; Pinay and Decamps, 1988; Burt and Haycock, 1993; Muscutt *et al.,* 1993). Thus, for example, Pinay and Decamps (1988) claimed that in a 30 m wide riparian forest all incoming nitrate from the uplands will be removed by processes of denitrification. Cooper (1990) demonstrated the very high potential of riparian soils for denitrification. Recently it has been claimed that denitrification processes are crucial for controlling the passage of nitrate through the riparian buffer zone while the role of vegetation remains unclear (Burt and Haycock, 1993; Haycock and Burt, 1993). However, one has to observe that good conditions for denitrification could promote release of other elements. Thus, for example, waterlogging of soils and anaerobic conditions could enhance phosphorus release (Hillbricht-Ilkowska *et al.,* 1995). It seems that the hydrology of element transfer through a buffer zone has an important bearing on such effects. Speed of groundwater passage is one of the most important hydrological factors (Lowrance *et al.,* 1983; Ryszkowski and Kedziora, 1993).

Nutrient uptake by plants is associated with evapotranspiration, but measurement of evapotranspiration rates under field conditions for whole stands of vegetation has met with various methodological problems. Recently, however, a method for evaluating the evapotranspiration rate of an intact vegetation stand has been developed using estimates of solar energy fluxes which drive evapotranspiration (Kedziora *et al.,* 1987; Ryszkowski and Kedziora, 1987; Kedziora *et al.,* 1989; Olejnik and Kedziora, 1991). Results of applying this method to the evaluation of the uptake of mineral nitrogen by plants within buffer zones, located in upland parts of a watershed, are presented in this paper. Additionally, mineral nitrogen transport through an unsaturated soil zone and long-term nitrogen storage in mid-field plantations are evaluated in order to get more information on internal nitrogen cycles in buffer zones located in upland parts of agricultural catchment areas.

MATERIALS AND METHODS

Study sites and climatic characteristics

The agricultural landscape in the vicinity of Turew has been the object of long-term studies on energy flow and matter cycling coordinated by the Research Centre for Agricultural and Forest Environments of the Polish Academy of Sciences. The area of interest is made up of slightly undulating ground moraine with many drainage valleys. Differences in elevation between higher and lower parts of the area do not exceed a few metres. In general, light soil (Hapludalfs and Udipsamments according to Soil Survey Staff, 1975) with favourable water infiltration conditions are found in the higher parts of the area. These loamy sands lie on sandy loam having much poorer infiltration rates. In the valleys Hapluquolls and Psammaquepts soils prevail. The cropping system of the studied agricultural landscape is characterised by a network of shelterbelts and mid-field plantations which were introduced in the 1820s. The period of plant growth on average lasts for 225 days (from March 21 to October 30). The period of active growth of plants on average begins on April 27 and ends on October 6 (167 days) when air temperatures are above +10°C (Wos, 1987). The winter, when the air temperatures are below O°C begins on average on December 12 and ends on March 3 (80 days).

Studies on water and chemical fluxes were carried out in 1984-1986 in a 65-year-old birch *(Betula verrucosa)* forest with some oak *(Quercus robur* and *Q. petrea)* and also in an adjacent field. Typic Hapludalfs soil constituted the field soil and under the forest the soils were Typic Udipsamments.

A 60-year-old pine *(Pinus silvestris)* forest and the adjoining cultivated field, both located on Typic Hapludalfs, were the object of a water and nitrate flux study in 1994. Small admixtures (<5%) of larch *(Larix europea)*, oaks *(Q. robur, Q. petrea)* and birch *(B. verrucosa)* trees were characteristic of the pine forest.

Five small watersheds (from 75 ha to 216 ha in area), having various extents of cover by shelterbelts (from 0.9 to 21%) and grasslands (from 0 to 27%), were selected for a study of nitrate output from watersheds variously structured with plant cover. Watersheds were chosen which possessed similar soil drainage rates in the cultivated fields. In watersheds selected for investigation, Typic Hapludalfs were found. They occurred with an admixture of Typic Udipsammens and Dystric Eutrochrepts and in total constituted from 42 to 97% of the watershed areas. The soil texture of the surface layers of these soils (up to 60-80 cm depth) was composed of loamy sand lying over sandy loam. The characteristic feature of these soils is fast drainage of surface horizons and medium to slow drainage of the deeper sandy loam horizons.

In valleys, as well as in local depressions, Histic Haplaquolls and Medisaprists were found, together constituting a few percent of the watershed area. In watersheds 1 and 2, located in the lower parts of the study region, significant parts (37% and 54%) are Typic Haplaquolls in which meadows or riparian forests can be found. The share of grasslands and forests in total varies from 0.9% to 48%. The watershed areas are drained by ditches and from their outlets samples for the analysis of nitrate contents were obtained every other week.

In a study of nitrogen storage in soil humus, the measurements were carried out in a 130-year-old shelterbelt (36 m wide) and in an adjoining field both located on Typic Hapludalfs. The shelterbelt is composed of black locust *(Robinia pseudoacacia)* with some oaks *(Q. robur)* and with elder *(Sambucus nigra)* and whitethorn *(Crataegus mongyna)* in the understorey.

The annual average air temperatures for the years studied varied from 8.1°C (1985) to 10.2°C in 1994 (Table 1). The warmest plant growth season (April-September), with a temperature of 16.8°C, was observed in 1994 and the coldest one, with a temperature of 13.9°C, in 1994. The last quarter of 1993 was much cooler (3.7°C) than the quarter preceding the 1984 year (Table 1).

Regarding annual precipitation, 1984 was the driest year with only 608 mm and was preceded by another very dry and warm year in 1983 (precipitation was only 500 mm, Table 1). The highest precipitation was observed in 1985 (797 mm). 1986 and 1994 received similar amounts of precipitation with amounts of about 760 mm and both these years were preceded by wet years with precipitation of about 800 mm.

Table 1. Air temperature and precipitation for the years of study.

	Period	*Air temp. (°C)*	*Precipitation (mm)*
1983	Total year	9.9	500
	October-December	4.3	122
1984	Total year	8.9	608
	April-September	13.9	452
1985	Total year	8.1	797
	April-September	14.8	572
1986	Total year	8.7	755
	April-September	15.6	463
1993	Total year	9.2	801
	October-December	3.7	194
1994	Total year	10.2	763
	April-September	16.8	445

Field treatments

In order to evaluate the dominant direction of subsurface water pathways in the birch and pine study sites, the elevation of the groundwater table was measured in wells distributed in irregular networks. On the basis of an annual survey (carried out at monthly intervals), the maps for each month of the year were drawn. Using this information, one well in the field and one inside the forest were chosen to indicate the dominant direction of subsurface water discharge. At both sites the wells in the field (P3 and S2) were located close to the forest edge to characterise discharges from the fields. The well (Z4) inside the birch forest was located at a distance of 50 m from its edge and, in the case of the pine forest, the well (S9) was located 56 m from the edge.

Samples of water for chemical analysis were taken when the well was refilled after water removal. In studies on five small watersheds the samples of water were taken from running water at the outlet of each watershed.

Soil samples for evaluation of organic carbon and nitrogen accumulation in shelterbelts were taken down to a depth of 30 cm, both in the forest and the adjoining field. In these comparisons neither humus loss, connected with tillage activity, nor the possibility of humus transfer from field into shelterbelt, by wind erosion, were taken into consideration. Thus the estimated values of organic carbon and nitrogen accumulation in forests should only be treated as rough estimates. Fertilisers were not applied to the forests.

a) The doses of N and P in mineral fertiliser applied to the field adjoining the birch forest were as follows: in 1984 a total of 56 kg ha^{-1} were spread as $(NH_4)_2SO_4$ and in 1985 a total of 78 kg ha^{-1} as NH_4NO_3. Total nitrogen input in 1986 was equal to 85 kg ha^{-1} (NH_4NO_3).

b) In 1994, 88 kg ha^{-1} of N were spread in the form of NH_4NO_3 over the field adjoining the pine forest.

c) The inputs of nitrogen in fertilisers applied in spring 1996 to the cultivated fields of the five small watersheds were from 86 to 116 kg ha^{-1}.

Laboratory treatments

Chemical analyses
Water samples were filtrated through Whatman GF/C glass filter paper (0.45 μm). Concentrations of nitrate and ammonium were measured colorimetrically using for:

- NO_3^- – brucine; NH_4^+ – Nessler's reagent.
- Organic C in soil was estimated by $K_2Cr_2O_7$ oxidising in a boiling solution of sulphuric acid.

Moisture saturation extracts of soil
Soil samples were collected separately from various layers of the unsaturated zone to the depth of 2.5 m. Soil was homogenised, sieved to remove roots and rocks and then treated with distilled water to saturation (Jackson, 1964). After 24 hours, water extracts were centrifuged from soil samples and concentrations of NO_3-N and NH_4-N were determined. A series of samples from various soil layers were taken in April and September of 1986 and in April of 1987 from nearby spots located in the cultivated field (close to well S2) and in the pine forest 56 m from its edge (close to well S9). In order to convert concentrations of mineral nitrogen, in the moisture saturation extracts of soil from a given layer, into estimates of nitrogen concentration in kg ha^{-1}, the following formula was used:

$C_l = (Q\ m\ g\ h) : 100$

where:

C_l is the mineral nitrogen (kg ha^{-1}) in a given soil layer

Q-N is the concentration in the extract (mg l^{-1})

m is the moisture content of the saturated soil sample (%)

g is the bulk density of the soil layer (kg m^{-3})

h is the depth of the soil layer (m)

METHODS OF ESTIMATION OF EVAPOTRANSPIRATION AND SUBSURFACE WATER FLUX

Evapotranspration as a measure of nutrient uptake

The uptake of chemical compounds dissolved in groundwater by plants is one of the most important factors controlling the dispersion of pollutants in groundwater in an agricultural landscape. In forests, water transpired during the plant growth season makes up more than 90% of the total evapotranspiration (Molga, 1986) because evaporation of water from shaded soil is low. Thus, evapotranspiration in forests can be regarded as a good index of the water uptake by root systems during the plant growth season. Because of sampling problems, it is difficult to have direct estimations of evapotranspiration rates for a total vegetation stand under field conditions. The problem can be solved by taking advantage of a basic physical phenomenon, that solar energy is the driving force for evapotranspiration. Thus, one can estimate the amount of evaporated water if the amount of energy used for this process is known. The heat balance quantifies the partition of intercepted energy, denoted by net radiation flux (Rn), into energy used for evapotranspiration (latent heat flux, LE), heating the air (sensible heat flux, A), soil heating (S) and other processes, which are driven by the small amounts of energy, such as production of plants. Meteorologists deal usually with the three first processes. A model for evaluating the heat balance of a plant stand was recently developed (Kedziora *et al.*, 1987; Ryszkowski and Kedziora, 1987; Kedziora *et al.*, 1989; Olejnik and Kedziora, 1991; Ryszkowski and Kedziora, 1993). The energy fluxes for evapotranspiration, air and soil heating are estimated by empirical equations relating climatological parameters to energy fluxes. The empirical equations were tested under field conditions in special experiments when latent and sensible heat fluxes were directly measured by the mean profile method (Kedziora *et al.*, 1987; Olejnik and Kedziora, 1991).

Calculation of evapotranspiration from climatological data

Climatological characteristics such as air temperature at 2 m height and soil temperature at 5, 10, 20, 50 cm depth, sunshine, wind speed, vapour pressure, water vapour saturation deficit, precipitation and humidity were measured by standard methods under field conditions. Using these direct measurements the elements of heat balance were estimated using the equations presented below.

The daily values of evapotranspiration of the vegetation were calculated from the heat balance equation:

$$LE = Rn - A - S \tag{1}$$

where:

Rn – net radiation flux (J m^{-2} day^{-1})

LE – latent heat flux (J m^{-2} day^{-1})

A – sensible heat flux (J m^{-2} day^{-1})

S – soil heat flux (J m^{-2} day^{-1})

Having calculated the flux of latent heat, the daily value of evapotranspiration (ETR) was calculated from the equation:

$$ETR = LE/L$$

where:

L is latent heat of vaporisation equal to 2.448×10^6 J kg^{-1}

ETR in this equation is expressed in mm, because one kg of water spread on the surface of one square metre gives a water layer 1 mm thick.

Individual components of heat balance were calculated in the way described below.

Net radiation was calculated according to the Penman formula:

$$Rn = (1-\alpha) \times Ro \times (0.22+0.54u) - \sigma(t+273)^4 \times (0.56-0.08\sqrt{e}) \times (0.10+0.9u) \quad (2)$$

where:

α – albedo of the study surface (dimensionless)

Ro – radiation on a plane surface at the outside of the atmosphere (J m^{-2} day^{-1})

u – relative sunshine (dimensionless)

t – air temperature (°C)

e – water vapour pressure (mbar)

σ – Stephan-Boltzman constant equal to 4.9×10^{-3} J m^{-2} day^{-1} K^{-4}

The daily soil heat flux during the growing season into soil is very small (Kedziora *et al.*, 1989; Kapuscinski and Moczko, 1990), only a small part of net radiation (from 2 to 7 percent of Rn). Because of difficulties in directly measuring soil heat flux, it was calculated from the following formula:

$$S = 0.05 \times Rn \quad (3)$$

The sensible heat flux was calculated from the formula:

$$A = \rho_a \times c_a \times h_s \times (T_a - T_s) \quad (4)$$

where:

ρ_a – air density (1.2 kg m^{-3})

c_a – specific heat of air (1004 J kg^{-1} K^{-1})

T_s – temperature of the evaporating surface [K]

T_a – temperature at 2 m above the active surface [K]

h_s – coefficient of turbulent exchange calculated from the following formula:

$$h_s - v \times k / [\{(\ln (z-d)/z_s) - f_s\} \times \{(\ln (z-d)/z_o) - f_m\}] \quad (5)$$

where:

v – wind speed 2 m above the active surface (m s^{-1})

k – von Karman's constant (0.41)

z – height at which measurements were taken (m)

d – zero plane displacement usually equal to 2/3 of the plant height h (m)

z_o – roughness parameter for momentum transfer (0.13 × h)

z_s – roughness parameter for heat transfer

where:

$$z_s = Z_o \times \exp(2) \text{ (Kanemasu } et\ al.\text{, 1979)} \quad (6)$$

h – plant height (m)

f_s, f_m – stability functions expressing the impact of atmospheric stability on heat and momentum transfer respectively (Kanemasu *et al.*, 1979)

$$f_s = f_m = 1-(1\text{-}5Ri) \text{ when } Ri > 0 \quad (7)$$

$$\ln f_s = 0.598+0.390 \times \ln(-Ri) - 0.090 \times (\ln(-Ri))^2 \text{ when } Ri<0 \quad (8)$$

$$\ln f_m = 0.032+0.448 \times \ln(-Ri) - 0.132 \times (\ln(-Ri))^2 \text{ when } Ri<0 \quad (9)$$

$$Ri = (g/T_a) \times [(dt/dz) \ / \ (dv/dz)^2] \quad (10)$$

where:

Ri = Richardson number (measure of stability)

g = acceleration due to gravity = 9.81 m s^{-2}

dt/dz; dv/dz = vertical air temperature gradients and wind speed gradients in the measurement layer.

Calculation of the daily values of evapotranspiration from the shelterbelt surface were carried out for the following conditions:

a) The height of trees (h_{max}) was 15-18 m.

b) At the meteorological station, air temperature (T_i) measurement is carried out at a height of 2 m above the ground surface covered with a few centimetres of short grass. To evaluate the temperature of the active surface (T_s) (the level of active surface depends on plant height), and air temperature at the level 2 m above the active surface (T_a) the following formulae were used:

$$Ts = T_i - b \times \ln(h+c) \quad (16)$$

$$T_a = T_i - b \times \ln(z+c) \quad (17)$$

Coefficients b and c were established on the basis of the following equations obtained from our own long-term investigations:

$$b = 0.5 \times \{1+\sin[(i-104) \times 2\pi/365]\} \quad (18)$$

$$c = 0.6 + 0.4 \times \cos[(i-95) \times 3.6\pi/365] \quad (19)$$

where i is ordinal number of day, starting from January the 1st.

The measurement of wind speed (v_i) at the meteorological station is carried out at a height of 15 m above ground. Wind speed (v) at a height of 2 m above the active surface was calculated as follows:

$$v = v_a \times \ln[(z-d)/z_o] \quad (20)$$

$$\text{where: } v_a = v_i/\ln[(z-d)/z_o] \quad (21)$$

Calculations of subsurface water flux

The water flux in a layer of soil having the width of 1 metre during a ten-day period may be calculated from the following formula (Drainage Principles and Applications, 1979):

$$J=10 \times K \times dh/dl \times D \quad (22)$$

where:

J is the ten-day water flux (m^3)

K = hydraulic conductivity (m day^{-1})

D = depth of filtration layer (m)

dh/dl = hydraulic gradient of groundwater table (m m^{-1})

The values of parameters characterising the water flux in the saturated zone of the studied landscape were as follows:

a) A hydraulic conductivity equal to 0.25 m day^{-1} under pine forest was estimated according to soil texture characteristic using methods provided in Drainage Principles and Applications (1979). The hydraulic conductivity under birch forest was estimated by Marcinek *et al.* (1990) to be equal to 0.5 m day^{-1}.

b) Mean depth of filtration soil layer is 2 m.

c) Distance between wells P3 (field) and Z4 in the birch forest is 50 m and between wells S2 and S9 (the pine forest) is 56 m.

Calculations of water and chemical compounds fluxes were carried out for a shelterbelt stretch 1 m wide.

The flux of chemical compound (M) taken up by shelterbelt between two wells during a given ten-day period was calculated by use of the formula:

$$M = J \times (C_1\text{-}C_2) \quad (23)$$

where:

M – flux of chemical compound (mg $m^{-1} \times$ ten $days^{-1}$)

J – water flux in ten-day period ($l^{-1} \times m^{-1} \times$ ten $days^{-1}$)

C_1– concentration of chemical compound in (mg l^{-1}) under field

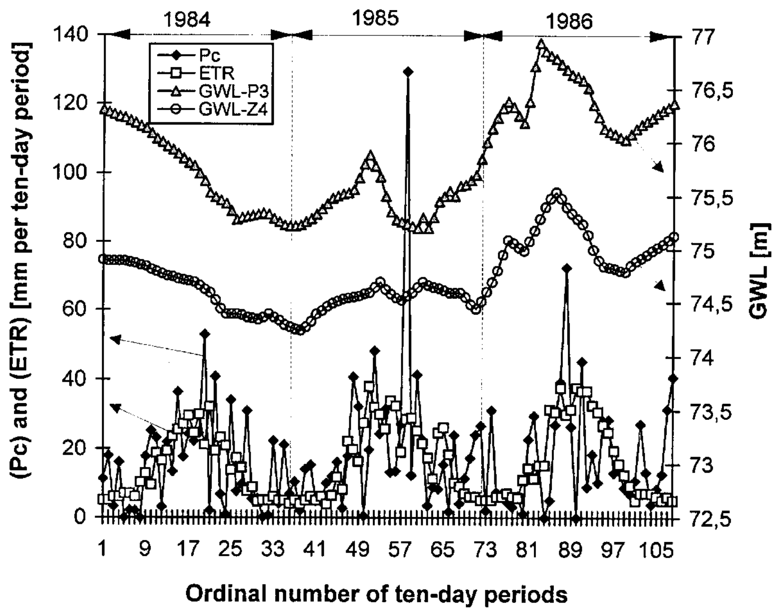
C_2– concentration of chemical compound in the forest (mg l^{-1}).

EVAPOTRANSPIRATION AND SUBSURFACE WATER FLUX

Intense transport of water from subsurface phreatic aquifer against the gravity gradient in an unsaturated zone of soil is only performed in plants possessing xylem systems. The upper layer quickly dries in soil devoid of plants; this stops evaporation of water from deeper layers (Kedziora, 1995). This is the reason why a fallow field conserves water. Thus evapotranspiration, which has an impact on a subsurface water flow, also influences the flux of chemical compounds dissolved in the phreatic aquifer. Mid-field forests or shelterbelts, having high roughness of canopy and low reflection of solar rays (low albedo), intercept higher amounts of incoming solar energy. Such woodlands tend to have higher net radiation values than cultivated fields or meadows (Ryszkowski and Kedziora, 1987). Trees also have well-developed root systems enhancing the uptake of groundwater from deep soil layers. Their canopy is exposed to higher wind velocities than canopies of cultivated plants or grasses. Interplay of these factors that influence evapotranspiration ensures that shelterbelts use a higher proportion of intercepted solar energy (net radiation) for evapotranspiration than cultivated fields or meadows. In studies on the heat and water balance of the Turew agricultural landscape, Ryszkowski and Kedziora (1987) estimated that during the plant growth season under typical weather conditions, the evapotranspiration in shelterbelts is higher by about 100 mm than those in meadows and about 150 mm higher than those in cultivated fields. However, in wet years, these differences can be smaller (by several tens of mm), while in dry years differences can be as much as 220 mm (Ryszkowski and Kedziora, 1995). Additionally, evapotranspiration rates can be much increased if there are higher temperatures over the adjoining field than in the shelterbelt canopy and if, above the shelterbelt, a temperature inversion is established. Then, wind blowing from the field carries advection heat energy to the tree canopy, enhancing the evapotranspiration rate (Ryszkowski and Kedziora, 1993). It has been shown that under such heat advection conditions on a sunny day (influx of solar energy of 100 W) subsurface water flow was reduced by 0.56 when the hydraulic gradient was 0.04 and water almost stopped flowing when the gradient was 0.01 (Ryszkowski and Kedziora, 1993). Thus one can expect great variation in subsurface groundwater fluxes depending on weather conditions.

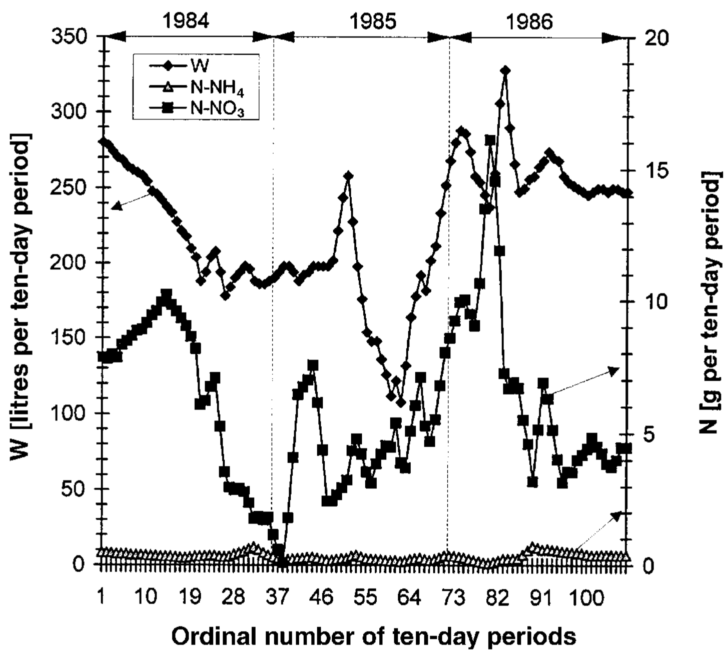
In the case of the study site the groundwater table was over 2.5 m below the ground surface and roots of cultivated plants in the field could not reach water in the phreatic zone. Thus cultivated plants with shallow root systems could only uptake water from moisture stored in the unsaturated soil zone. The well-developed, deep root system of trees enables water uptake from the phreatic aquifer. The decrease of moisture in an unsaturated zone does not influence the evapotranspiration rate of trees very much because the roots have access to flowing water in the saturated zone. Thus trees can easily influence the chemical constitution of the phreatic zone water. The mean value of groundwater level (a.s.l.) under the field was 75.8 m, and 50 m from the forest edge it amounted to 74.7 m, 1 m lower than in the field. The variation in groundwater level under field ranged from 75.2 m a.s.l. to 76.9 m a.s.l. (Fig. 1); the coefficient of variation (cv) was 0.66%. In the forest the groundwater level ranged from 74.2 m to 75.6 m, with a cv of 0.46%. The variability of groundwater level was 40% greater in the field than in the forest. The lowering of groundwater level in 1984 was a result of the soil drying up in the previous two years (Fig. 1). Relatively high rainfall in 1985, amounting to 797 mm, contributed to gradual reconstruction of water supplies in the soil.

Figure 1. Changes of precipitation (Pc), groundwater level (GWL) under the field (P3) and birch forest (Z4), as well as evapotranspiration (ETR) in the forest.



On the basis of hydraulic calculations (equation 22), water discharge from the field to the forest was estimated (Fig. 2). Through a plane of the phreatic aquifer 1 m wide and 2 m deep, 8.0 m^3 of water flowed from the field into the forested area in 1984. In 1985, the amount of water passing was equal to 6.7 m^3 and in 1986 – 9.4 m^3. The increasing amounts of water were a result of higher rainfall. Evapotranspiration in the forest amounted to 640 mm in 1984, 550 mm in 1985 and 610 mm in 1986 (Fig. 1). Thus the input of groundwater from the field is in the range of 5%-15% of water evaporated from 1 ha of this forest.

Figure 2. Flux of water (W), NO_3-N and NH_4-N through a phreatic zone under the birch forest.



The subsurface water flux was studied under the cultivated field and the pine forest during a wet year (annual precipitation in 1994 was equal to 763 mm) which followed after a very wet year (precipitation in 1993 was equal to 801 mm). The average elevation of the water table under the field was 82.5 m a.s.l. with a coefficient of variation equal to 0.66% which corresponded to a range of water table changes from 81.9 m a.s.l. to 83.6 m a.s.l. (Fig. 3). The mean elevation of the water table under the forest was 80.1 m a.s.l. with a coefficient of variation of 0.54% which reflected changes in the water table from 80.0 m a.s.l. to 81.2 m a.s.l. Estimated subsurface water export from the field to the pine forest, through a 1 m x 2 m plane, amounted to 5.0 m^3 of water (Fig. 4). Evapotranspiration in 1994 was 569 mm.

Figure 3. Changes of precipitation (Pc), groundwater level (GWL) under the field (S2) and pine forest (S9), as well as evapotranspiration (ETR) in the forest.

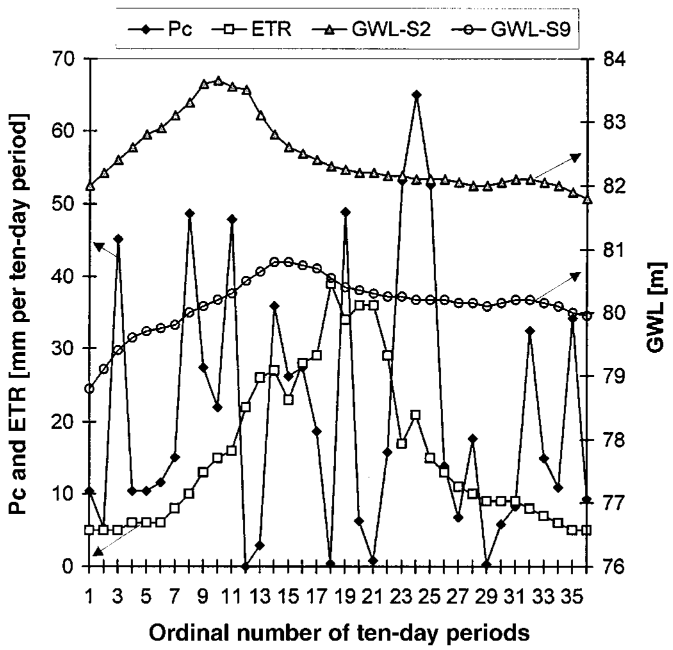
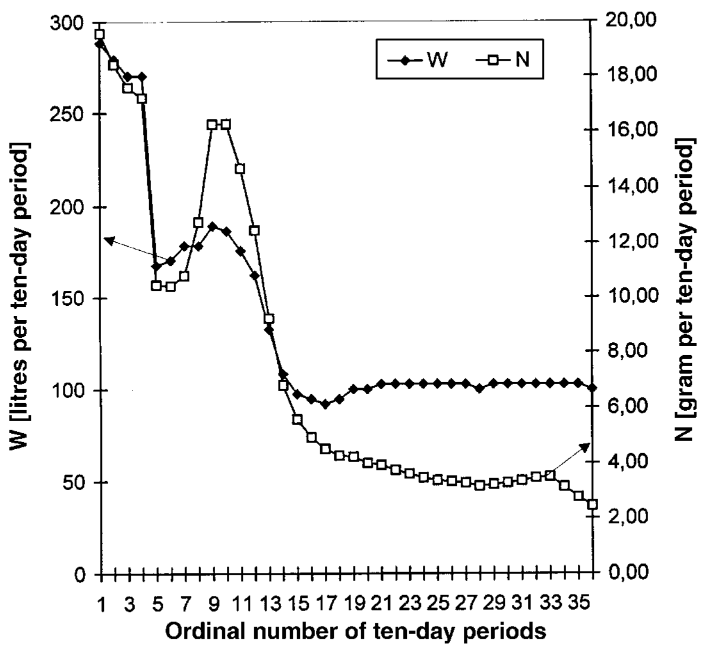


Figure 4. Flux of water (W), NO_3-N (N) through a phreatic zone under the pine forest.



There is a complicated relationship between rainfall intensity and changes of groundwater level. For example, very heavy rain during the summer of 1985 (Fig. 1) had no impact on groundwater level because much of the precipitation was lost in surface runoff during this short event. Thus the highly sporadic nature of precipitation is not reflected by changes in groundwater levels (Figs. 1 and 3). Because of the complicated interplay of factors influencing interception of precipitation, there is also a poor correlation between annual precipitation and the magnitude of water discharge through the phreatic zone. Thus the highest annual precipitation (797 mm) was observed in 1995, while the highest subsurface water discharge was in 1986 when the effects of two previous wet years had accumulated. The evapotranspiration rates show the most regular cycles with peaks in summer months despite of irregular patterns of precipitation.

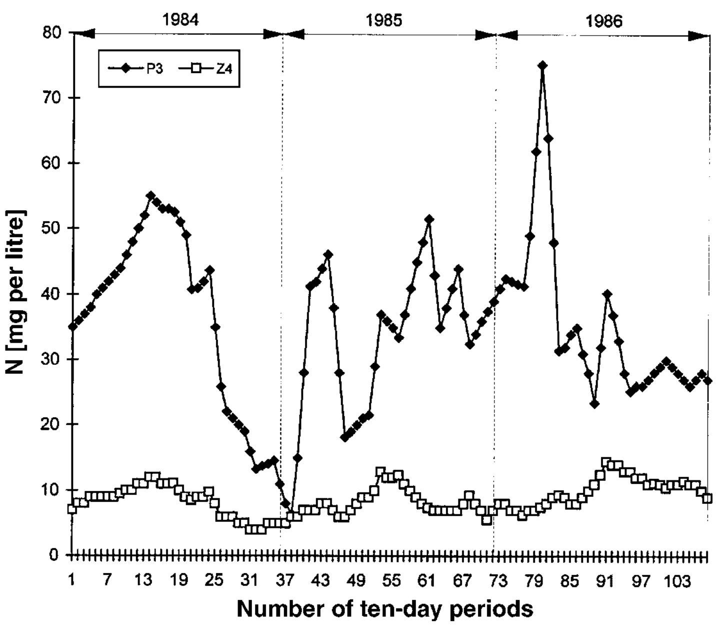
CONTROL OF NUTRIENT FLUXES IN GROUNDWATER BY MID-FIELD WOODLAND

Birch forest

In the three-year period, the mean NO_3-N concentration was 33.1 mg l^{-1} in the field and 8.4 mg l^{-1} in the forest (Fig. 5). The variation in NO_3-N concentration was higher in the field phreatic zone (cv=44%) than in the forest (cv=32%). The annual flux of NO_3-N with discharging groundwater from the field into the birch forest through the 1 m x 2 m phreatic plane was 301 g in 1984, 219 g in 1985 and 335 g in 1986 (Fig. 2).

The mean NH_4-N concentration for the three years was 1.73 mg l^{-1} (cv=32%). For the same years discharges of NH_4-N from the cultivated field into the birch forest were 14.2, 8.6 and 13.3 g respectively (Fig. 2). Total input of NH_4-N during the three years was equal to 36 g while the output was 41 g.

Figure 5. Changes of NO_3-N (N) concentrations in groundwater under the field (P3) and the birch forest (Z4).



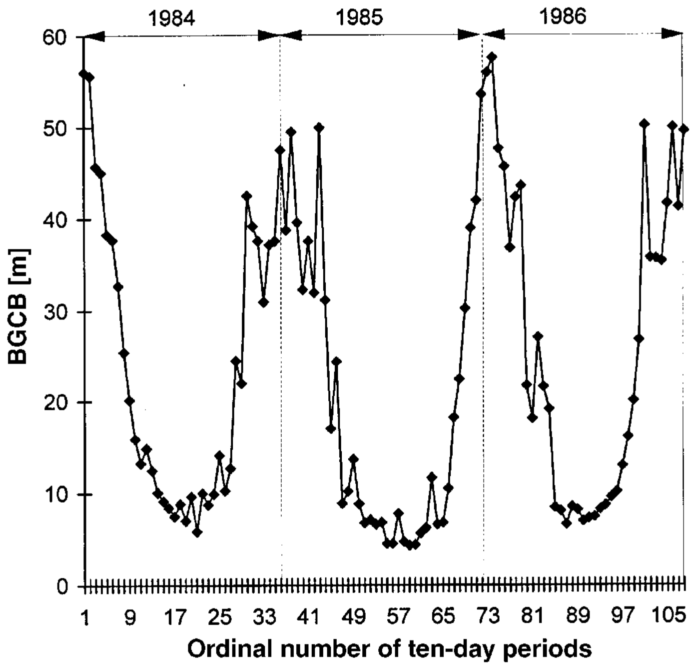
Using the evapotranspiration rates (Fig. 1) and the amounts of water discharged into the birch forest from the field (Fig. 2), the length of one metre wide forest needed to evaporate the water can be estimated. It is presumed that nutrients dissolved in this water are uptaken by plants too. In these calculations it is assumed that the uptake of nutrients is characterised by mass flow and effects of the absorption by diffusion processes are neglected. The calculations of the biogeochemical barrier (BGCB) length were done in the following way.

For example, in a ten-day period between 20 and 30 April, 246 litres of water flowed from the field into the forest through a 1 m x 2 m plane of the phreatic zone. In this ten-day period evapotranspiration was equal to 25 litres per m^2, so the amount of water equivalent to, but not necessarily the same in-flow water, will be transpired by trees in an area of 1 m x 10 m. Thus at a distance of 10 m nutrients contained in influx water were also removed.

It should be emphasised that water flow is very slow in soil and in the case analysed it amounts to about 1-2.5 m per 24 hours. For greater fluxes (e.g. higher water table slope) more water will be passing and the effectiveness of a given length of the BGCB will be smaller.

In the case of the birch forest the width of BGCB changed during the plant growth period from 5 to 25 m (Fig. 6) with an average value of 10.4 m. Thus one can assume that a birch BGCB of 20 m wide will efficiently reduce the inflow of nitrate from the field. In deciduous trees, when the leaves are shed in the late autumn evapotranspiration stops, while evapotranspiration in coniferous trees either stops or is highly depressed because of unfavourable climatic conditions in the period of late autumn-winter. Thus in the winter season the evaporation of water from the earth surface is the dominating process and calculation of BGCB width makes no sense.

Figure 6. Changes of biogeochemical barrier (BGCB) length in the birch forest.



In addition to NO_3-N , ammonium (NH_4-N) was determined in the groundwater. In the three years the concentration of NH_4-N was higher in the phreatic aquifer under the birch forest than under the adjoining cultivated field, but only in 1985 were the differences statistically significant (Table 2). One can assume that there is balance between uptake of NH_4-N by plants and its release in processes of mineralisation and denitrification. Nevertheless, comparing the concentrations of both mineral forms of nitrogen (nitrate + ammonia), one finds substantially lower concentrations in groundwater beneath the forest (10.51 mg l^{-1}) than in the phreatic aquifer below the cultivated field (36.64 mg l^{-1}).

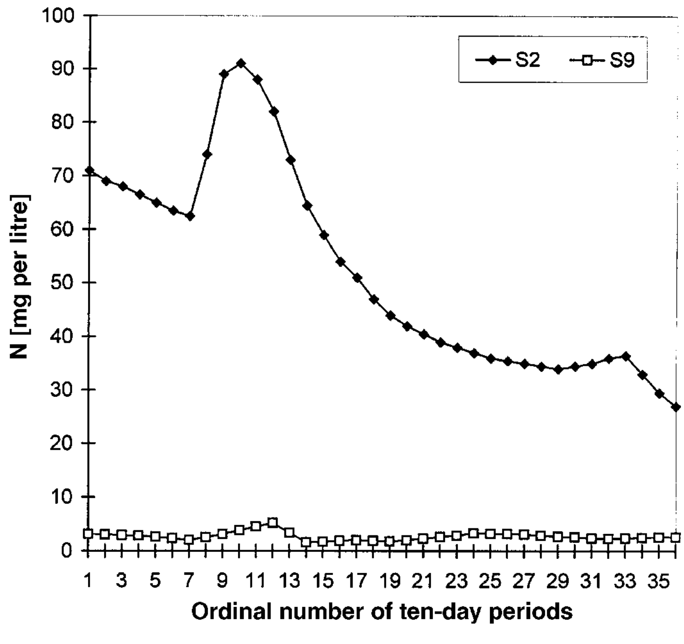
Table 2. Concentration of NH_4-N (mg l^{-1}) in groundwater under fields and forests.

Year	*Field*	*Forest*	*Statistical significance*
1984	1.86	1.96	no
1985	1.28	1.77	$p < 0.05$
1986	1.43	1.44	no
1994	1.27	1.07	no

The pine forest

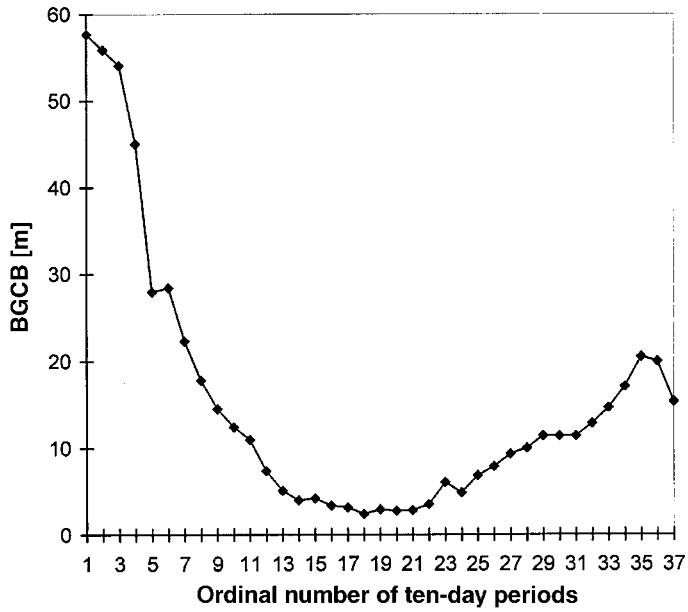
There was a steep gradient in NO_3-N concentration in groundwater between the field and the pine forest from a mean value of 52.4 mg l^{-1} to 2.7 mg l^{-1} (Fig. 7). The coefficient of variation of NO_3-N concentration in the phreatic aquifer below the cultivated field was 36.3%, while below the forest it was lower (27.6%), resembling the situation observed in the cultivated field and the birch forest. The annual flux of NO_3-N through a 1 m x 2 m plane of the phreatic aquifer at the forest boundary was equal to 287 g. The highest rates of NO_3-N inputs were detected in the period between January and April which was the result of very high precipitation in autumn 1993 and a warm winter (Table 1). In January, NO_3-N discharges from the field reached the level of almost 20 g per ten-day period before decreasing (Fig. 4). After heavy rain (48 mm) in February (Fig. 3) when the unsaturated zone of soil was very moist, increased inputs of NO_3-N into the pine forest reached almost 16 g N per ten-day period (Fig. 4). When evapotranspiration reached a rate of 25 mm per ten days (Fig. 3), the water flux stabilised at a level of 100 litres per ten days and NO_3-N flux slightly decreased as the season progressed (Fig. 4).

Figure 7. Changes of NO_3-N (N) concentrations in groundwater under the field (S2) and the pine forest (S9).



During the growing season of 1994 the width of biogeochemical barrier (BGCB) varied from 2.4 m to about 10 m with an average value of 5.8 m (Fig. 8). The mean value of BGCD is lower in the pine forest than in the birch forest; the difference is statistically significant. This may be an indication of the higher efficiency in nitrate flow control by a coniferous forest than by a deciduous one, but it should be kept in mind that the birch forest in question was a poor stand of trees.

Figure 8. Changes of biogeochemical barrier (BGCB) length in the pine forest.

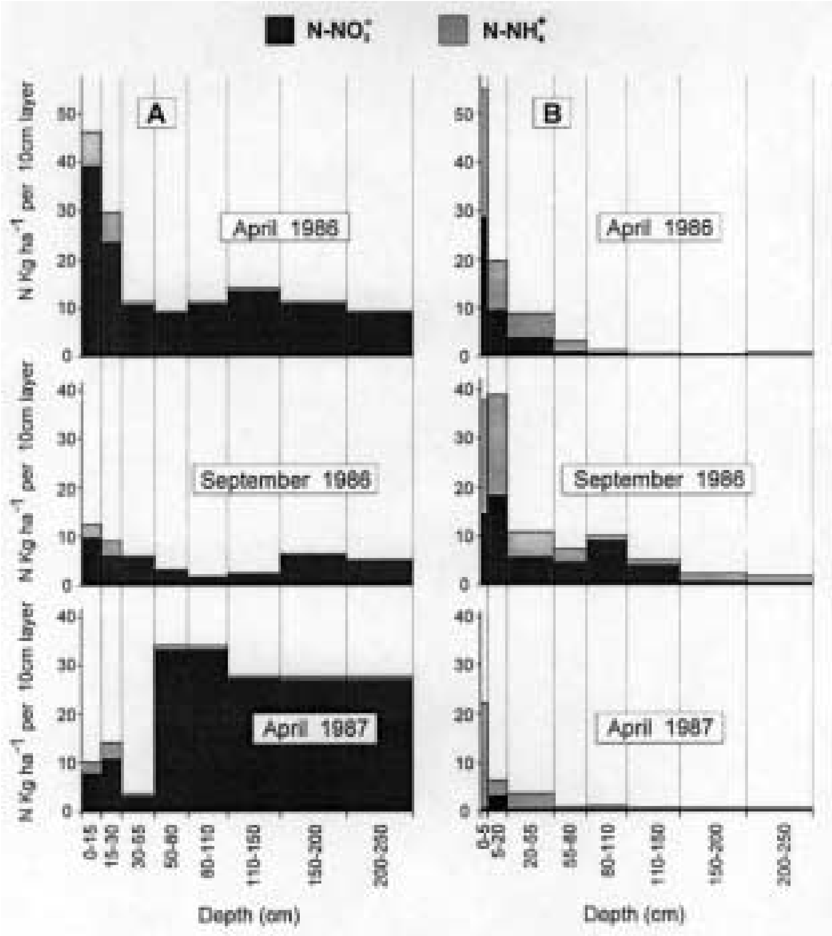


The NH_4-N concentration in subsurface water inside the pine forest amounted to 1.07 mg l^{-1} and was not statistically different from those under the field (Table 2). The annual influx of NH_4-N into the forest was equal to 7.8 g and output after groundwater passed under trees was equal to 6.7.

THE DISTRIBUTION OF MINERAL FORMS OF NITROGEN ACROSS THE UNSATURATED ZONE OF THE SOIL PROFILE IN THE CULTIVATED FIELD AND THE PINE FOREST

A completely different cycle of mineral nitrogen (NO_3^-N + NH_4-N), in time and depth, was detected under the cultivated field and the pine forest (Fig. 9). The highest amounts of total mineral N under the field, up to 2.5 m depth, were found in the spring of 1986 (360 kg ha^{-1}) and spring 1987 (616 kg ha^{-1}) with a low of 127 kg ha^{-1} in September 1986. For the same profile under the pine forest, the peak of mineral nitrogen (182 kg ha^{-1}) was observed in September 1986 with lows in the spring of 1986 (97 kg ha^{-1}) and the spring of 1987 (43 kg ha^{-1}) (Fig. 9). Beside this difference one can point out that the mean content of N under the cultivated field (368 kg ha^{-1}) is more than three times higher than under the pine forest (107 kg ha^{-1}). Amplitude of the change is also greater in the unsaturated zone of the field. The other striking difference is the much higher contribution of NO_3-N to the pool of mineral nitrogen (Fig. 9). The overall mean NO_3-N content in soil under the field is 342 kg ha^{-1} while the contents of NH_4-N was 27 kg ha^{-1}. In contrast, it was found that the nitrate-nitrogen content was slightly lower in the unsaturated zone of the pine forest (53 kg ha^{-1}) than the content of NH_4-N (55 kg ha^{-1}). The rhythm of mineral nitrogen changes under the forest is associated with decomposition of shed plant biomass and leaching out of nitrogen from living tissue, litter and soil (Chapin, 1980; Cole and Rapp, 1981; Chapin and Kedrowski, 1983). In deeper layers of the unsaturated zone (below 150 cm of depth) the mineral nitrogen concentration is very low (1.61 mg l^{-1}) and the ratio of NO_3-N to NH_4-N varies from 0.28 to 1.00. Very low concentrations of mineral nitrogen below a depth of 150 cm indicate that the root system of trees effectively constrain nitrogen leaching down the soil profile. The ratio of NO_3-N to NH_4-N in groundwater under the forest was 7.8 and much lower than in the cultivated field where it was 26.3. This difference can be interpreted as follows: inside the forest, mineral nitrogen migrating with the groundwater originates from internal cycling within the forest soils and not from inputs from the field.

Figure 9. Distribution of mineral nitrogen (NO_3-N and NH_4-N) in the unsaturated layer of soil in the cultivated field (A) and pine forest (B).



The natural rhythm of higher leaching rates in autumn and winter observed in the forest is distorted by fertilisation in the cultivated field. The high level of mineral nitrogen in the unsaturated zone in spring is associated with fertilisation causing large quantities of nitrogen to be found in the deepest layers. A dose of 86 kg N ha^{-1} spread as NH_4NO_3 was applied for maize cultivation in 1986 before the samples of soils for moisture saturation extracts were collected (April 17th, 1986). Then in May, a dose of 133 kg N ha^{-1} (18 kg as $[NH_4]_3PO_4$ and 115 kg N ha^{-1} as $CO[NH_4]_2$) was applied. In the spring of 1987, before the samples were collected, input of N as NH_4NO_3 amounted to 85 kg.

The upper soil layers (up to 80-100 cm of depth) are loamy sand with good infiltration rates while deeper layers are composed of sandy loam with lower percolation rates. Quite high amounts of mineral nitrogen, especially in the form of nitrates, appear in the deeper soil layers (below 110 cm) (Fig. 9). These are outside the reach of maize roots and are thus in the process of leaching down to the saturated zone. High amounts of mineral nitrogen in spring, especially nitrate in the deeper layers of the unsaturated zone, indicate that the leaching is associated with higher percolation rates observed usually during late autumn to early spring. Analyses of mineral nitrogen distribution in the unsaturated zone below the cultivated field have shown that large amounts are leached into the groundwater. This is the result of a much more open N cycle and probably substantial amounts of fertiliser nitrogen are never incorporated into the biological processes of the agroecosystem.

THE INFLUENCE OF PLANT COVER STRUCTURE ON NO_3-N CONCENTRATIONS IN WATER OUTPUTS FROM SMALL WATERSHEDS

In the winter season (October to first half of April), apart from the period when water was frozen (middle of January to end of March 1986), six sets of samples were taken in five watersheds. In the summer season (middle of April to end of July 1986), two very intense rainfalls occurred. During the first ten-day period of May precipitation amounted to 106.4 mm and in the first ten days of July rainfall was 111.1 mm. Two sets of water samples associated with these precipitation events were analysed separately because of the appearance of intensive surface flow. The remaining six samples from the summer season were analysed separately. In all three situations a statistically significant correlation between the amount of buffer zone (forest and meadows) in a watershed and NO_3-N concentration ($p<0.01$) was found, with correlations of $r = -0.91$ ($r^2 = 0.084$) for the winter season, $r = -0.86$ ($r^2 = 0.74$) for the summer season and for the high rainfall period $r = -0.89$ ($r^2 = 0.79$).

A detailed analysis of the form of the relationship between NO_3-N concentration and buffer zone area (Fig. 10) showed that this relationship could be described well by an exponential function in form of the regression equation:

$$\ln C = \ln a + bP$$

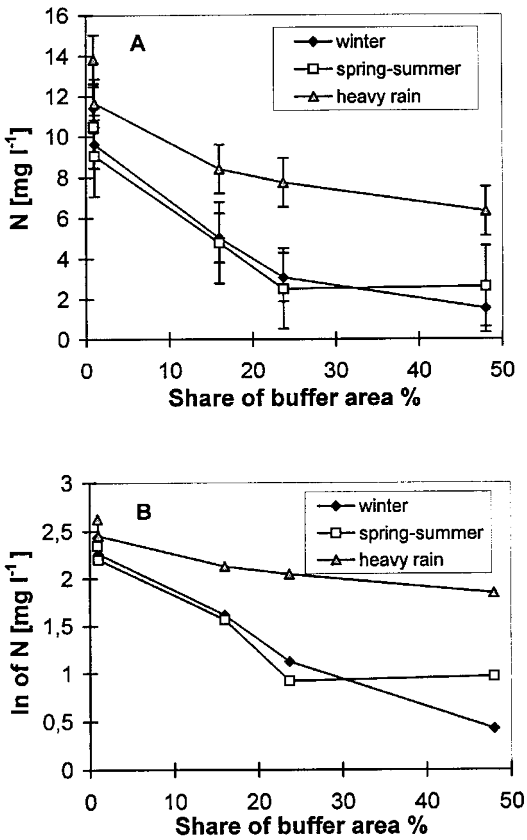
where:

P is the fraction of buffer zone area in the whole watershed

C is the NO_3-N concentration (mg l^{-1})

a and b are the regression parameters.

Figure 10. Influence of plant cover structure on NO_3-N concentration in water output: normal (A) and exponential relationship (B).



The regression equation for the winter season ($\ln C = 2.325 - 0.042P$) and for the spring-summer ($\ln C = 2.142 - 0.030P$) do not differ statistically; however, a difference occurs when compared with that calculated for the period of heavy rainfall ($\ln C = 2.492 - 0.015P$). Although there is still a significant correlation between NO_3-N concentration and the buffer zone area, there is little difference between the watersheds in winter and in spring-summer periods under normal rainfall (Fig. 10). The increased surface runoff during heavy rainfall may be the cause of the greater difference seen at such times. Concentrated surface runoff may reduce the efficiency of nitrate control by buffer zones, especially when channelised surface flow is created due to very intense precipitation (Muscutt *et al.*, 1993). The exponential relationship between nitrate concentration and the fraction of buffer zone in a watershed allows us to estimate how large an area should be covered with buffer zones in order to maintain the proper level of NO_3-N concentration. So, for example, for the winter season, defined by the function $C = 10.23e^{-0.042P}$, in a situation when cultivated fields have Hapludalf or Udipsamment soils, then 5% of buffer zones in the total area should maintain the concentration of NO_3-N in channels draining small watersheds at a level not higher than 8 mg l^{-1}. In a situation when the buffer zone area increases to 17% the concentration of nitrate should not exceed, under normal rainfall conditions, 5 mg of NO_3-N in a litre. In the analyses presented above, the probable impact of buffer zone location in regard to drainage channels was not taken into account, which can be of some importance in the case of large watersheds of several thousand hectares (Johnston *et al.*, 1990).

Long-term store of nitrogen in shelterbelts

The accumulation of soil organic matter under shelterbelts is the main mechanism of long-term withdrawal of various elements from the water moving through an agricultural watershed. Absorbed by trees, nutrients from groundwater become built into plant biomass and then, after the decay of plant tissues, some of them will be stored in humus as a result of biochemical processes taking place during plant decomposition. In order to estimate the accumulation of organic matter in forest soil, a comparison was made between the contents of organic carbon in the soil of the cultivated field and the pine forest, the pseudo-accacia shelterbelt and the birch forest.

In all three cases significant accumulation of humus was found under the forest (Table 3). Because of different relations of C:N in soils of the fields and forests, the amount of organic nitrogen stored is relatively smaller in forest soils than it could be concluded from the organic matter accumulation. Nevertheless, substantial amounts of nitrogen are stored in the humus of shelterbelts or mid-field forests (Table 3). Nitrogen stored in humus constitutes a part of the cycling nitrogen in the shelterbelt soil that is withdrawn for a long time. In the case of nitrogen stored in soil organic matter of the birch forest there is a difference in soils under the field and forest. In Udipsamments soils (sandy soils) formation of humus is more difficult due to small water storage in these soils and the small amount of mineral colloids. So, if this forest had been Hapludalfs soils, then more organic matter accumulation would probably have been detected.

Some nitrogen is also withdrawn from circulation for a considerable time, in woody parts of trees. The overstorey bole wood in the 60-year-old pine forest was 12430 kg ha^{-1}. Assuming the content of N in bole wood to be 0.0011 (Cole and Rapp, 1981) one can estimate the mass of nitrogen in boles of the pine stand to be equal to 13.60 kg ha^{-1} (or 1.3 g N m^{-2}). Dividing this value by 60 years, one can estimate the mean annual rate of N incorporation into wood to be equal to 0.022 g N m^{-2}. This rough estimate indicates that the annual withdrawal of nitrogen into boles is about 1.6% of the annual storage of N in humus (1.4 g m^{-2} y^{-1}) (Table 3). One has to remember that wood production was low in the forest studied, because the stock density was only 50% of the optimal value. The same calculation for birch forest gave the following estimate: because of low stock density the overstorey bole wood was 5850 kg ha^{-1}. Assuming nitrogen contents in bole wood birch to be 0.0014 (Cole and Rapp, 1981), the standing crop of N in bole wood was 8.19 kg ha^{-1} (0.819 gm^{-2}). In the situation of strongly cleared birch forest (about 25% of the optimal stock density), the annual storage of N in wood was 0.012 g m^{-2} y^{-1}, which is a slightly lower rate than that observed in the case of pine forest. In relation to N stored in soil organic matter, the annual withdrawal of N in wood is only 1%.

Table 3. Accumulation (kg ha^{-1}) of carbon and nitrogen in the surface layer of soil (up to 30 cm deep) under cultivated fields and shelterbelts.

Structures	*Pine Forest*	*Pseudo-accacia shelterbelts*	*Birch Forest*
Soil catena	*Hapludalfs - Hapludalfs*	*Hapludalfs - Hapludalfs*	*Hapludalfs - Udipsamments*
Age of trees (yrs)	*60*	*130*	*65*
Carbon			
Field (F)	23400	20700	19350
Forest (A)	43650	54900	37400
A-F	20250	34200	18050
Annual rate of accumulation	338	263	278
Nitrogen			
Field (F)	3150	2700	2250
Forest (A)	4005	4950	2940
A-F	855	2250	690
Annual rate of accumulation	14	17	11

DISCUSSION AND CONCLUSIONS

Studies of groundwater hydrology and chemistry in birch and pine mid-field forests have confirmed once more that a plant cover structure composed of cultivated fields and patches of permanent vegetation can control phreatic zone pollution. The studies on the influence of plant structure of the watershed area on NO_3-N concentrations at the outlets of draining ditches support this conclusion and, additionally, permit NO_3-N concentrations to be forecast, if the fraction of buffer zone area is known for the catchment. The much smaller ratio of NO_3-N to NH_4-N in the phreatic aquifer below the forest than below the cultivated field indicates that nitrate found in groundwater inside a forest originates from the internal N cycle in this ecosystem. If this interpretation is true then total influx of nitrate from the field is removed in less than a 50 m band of forest. Evapotranspiration estimations under field conditions allowed the length of buffer zone, within which an equivalent of the amount of water discharging from field is removed in evapotranspiration, to be calculated. This band is called the length of the biogeochemical barrier (BGCB). The value of the BGCB in the birch forest was estimated during the growing season to be 11 m and in the pine forest the BGCB length is equal to 6 m. The average length of BGCB in the birch forest was estimated from data for three consecutive vegetation seasons having different precipitation (Table 1). If only the 1986 data are used, which is climatologically similar to 1994 when studies in the pine forest were performed, then the width of the BGCB in the birch forest is 11.5 m; slightly greater than for pine. This result indicates that the type of vegetation could have an effect on the rate of chemical removal from groundwater.

The values reported in literature concerning buffer length required to remove subsurface inputs of nitrate range from 5 m to 30 m. Individual values were as follows: 5 m (Cooper, 1990), 8 m (Haycock and Burt, 1991), 16 m (Jacobs and Gilliam, 1985), 19 m (Peterjohn and Correll, 1984) and 30 m (Pinay and Decamps, 1988). There is a good match between estimates presented in this study and those reported in the literature. The estimations of BGCB length obtained in the Turew landscape studies fluctuate for various ten-day periods from 3 m to 25 m (Figs. 6 and 8). In other studies (Ryszkowski and Kedziora, 1993), it was found that very high transpiration (5.8 l day^{-1} m^{-2}) in a shelterbelt 10 m wide, caused by advection of heat energy from the field to the tree canopy, could practically stop all passage of shallow groundwater. Thus, changes in the energy available for evapotranspiration could cause rapid changes in transpiration and, by that token, influence uptake of nutrients from the groundwater. The results presented above suggest that performance of upland forests could act in a similar way to buffer strips on the floodplain. This similarity appears despite the fact that the main controlling function of buffer zones in the floodplains is assigned to the denitrification process (Pinay and Decamps, 1988; Jacobs and Gilliam, 1985; Lowrance *et al.,* 1985; Cooper, 1990; Burt and Haycock, 1993; Muscutt *et al.,* 1993), while in upland forests the main process is the uptake of nitrate by plants in water mass flow for evapotranspiration. To understand the interplay of denitrification processes and nutrient uptake by plants in control of nitrate passage in subsurface water flow through a buffer zone, one would have to carry out simultaneous study on the efficiency of these processes.

The direct estimation of nitrogen uptake by isolated roots of pine, oak and birch trees resulted in estimates of annual uptake of 9.1 g N m^2 (60-65% of it as NO_3-N) in the forests studied (Prusinkiewicz *et al.,* 1990). The studies were carried out in the Turew landscape and the afforestation was located on Alfic-Udipsamment soils.

Birch trees in our studies absorbed on average 215 g N y^{-1} within the biogeochemical barrier, which had 11 m length, and uptake by trees from 1 m^2 amounted to 20 g y^{-1}. These uptake rates are about 2 or 3 times higher than data from the papers cited above. In pine forest, uptake of nutrients, estimated by mass flow of water in evapotranspiration, was 270 g y^{-1} within the BGCD which is equal to 45 g m^{-2} y^{-1}, that is, about 5 or 6 times more than the values found in the cited papers.

It seems that withdrawal of nitrogen from subsurface water into long-term storage in the soil organic matter and in the overstorey bole wood is small in comparison to the dynamic withdrawal resulting from internal cycling. Annual withdrawal of N into organic matter was estimated to be 1.4 g m^{-2} y^{-1} in the pine forest and 1.1 g m^{-2} y^{-1} in the birch forest. The annual retention of NO_3-N and NH_4-N in pine forest was 274 g. The estimated width of the biogeochemical barrier needed was 6 m. Thus, annual storage of N in soil organic matter in the band of forest equal to the BGCB width, was 8.4 g y^{-1} while NO_3-N + NH_4-N retention was 274 g; the amount annually withdrawn is 3%. The same calculations for the birch forest resulted in estimation of the total retention of NO_3-N + NH_4-N to be equal to 211 g, while nitrogen stored in soil organic matter of the BGCB was 12.0 g; therefore the N withdrawn annually from internal cycling is 5.6%. The storage in bole wood makes up about 1% of the withdrawal in soil organic matter. Thus, about 94 to 97% of N removed by the buffer zone from subsurface water flow is incorporated in dynamic internal cycles composed of nutrient retranslocation, release in decomposition and reuse by root systems and other biological processes, as well as in cation exchange (NH_4^+), released in denitrification or NH_3 volatilisation, and so on. We can hypothesise that removal of mineral nitrogen from the groundwater is responsible for plant uptake while release from the ecosystem is achieved via the process of denitrification. From the point of view of protection of water reservoirs against nitrate pollution, one should stimulate the process of denitrification in practising buffer zone management.

REFERENCES

Baker, L.A. (1992) Introduction to nonpoint source pollution in the United States and prospects for wetland use. *Ecological Engineering,* **1**, 1-26.

Bartoszewicz, A. (1990) Chemical compounds in groundwater of agricultural watershed under soil climatological conditions of Koscian Plain (in Polish). In: Ryszkowski, L., Marcinek, J. and Kedziora, A. (Eds.) *Obieg wody i bariery biogeochemiczne w krajobrazie rolniczym.* Adam Mickiewicz University Press. Poznan. pp 127-142.

Bartoszewicz, A. (1994) The chemical compounds in surface waters of agricultural catchments under the soil weather conditions of the Koscian lowland (in Polish). *Roczniki Akademii Rolniczej.* Poznan. **25**, 5-68.

Burt, T.P. and Haycock, N.E (1993) Controlling losses of nitrate by changing land use. In: Burt, T.P., Heathwite, A.L. and Trudgill, S.T. (Eds.) *Nitrate: processes, patterns and management.* Wiley and Sons. Chichester. pp 341-367.

Chapin, F.S. III. (1980) The mineral nutrition of wild plants. *Annu. Rev. Ecol. Syst.,* **11**, 233-260.

Chapin, F.S. III and Kedrowski, R.A. (1983) Seasonal changes in nitrogen and phosphorus fraction and autumn retranslocation in evergreen and deciduous taiga trees. *Ecology,* **64**, 373-391.

Cole, D.W. and Rapp, M. (1981) Elemental cycling in forest ecosystems. In: Reichle, D.E. (Ed.) *Dynamic properties of forest ecosystems.* Cambridge Univ. Press. Cambridge. pp 341-409.

Cooper, A.B. (1990) Nitrate depletion in the riparian zone and stream channel of a small head water catchment. *Hydrobiologia*, **202**, 13-26.
Drainage Principles and Applications (1979) Inst. for Land reclamation and improvement. Wageningen. The Netherlands. Vol. **II**: 374 pp. and Vol **III**: 364 pp.
Haycock, N.E. and Burt, T.P. (1991) The sensity of rivers to nitrate leaching. The effectiveness of near-stream land as a nutrient retention zone. In: Thomas, D.S.G. and Allison, R.J. (Eds.) *Landscape sensivity.* Wiley and Sons. London. pp 261-272.
Hillbricht-Ilkowska, A., Ryszkowski, L. and Sharpley, A.N. (1995) Phosphorus transfers and landscape structure: riparian sites and diversified land use patterns. In: Tissen, H. (Ed.) *Phosphorus in the global environment.* Wiley and Sons. Chichester. pp 201-228.
Jackson, M.L. (1964) *Soil chemical analysis.* Prentice Hall, Englewood Cliffs N.J. USA. pp 498.
Jacobs, T.C. and Gilliam, J.W. (1985) Riparian losses of nitrate from agricultural drainage waters. *J. Environ. Qual.,* **14**, 472-478.
Johnston C.A., Detenbeck, N.E. and Niemi, G.J. (1990) The cumulative effect of wetlands on stream water quality and quantity. *Biogeochemistry,* **10**, 105-141.
Kanemasu, E.T., Wesely, M.L. and Heiman, J.L. (1979) Techniques for calculating energy and mass fluxes. In: Barfield, B.J. and Gerber, J.F. (Eds.) *Modification of the aerial environment of plants.* American Society of Agricultural Engineers. St. Joseph, Michigan USA.
Kapuscinski J. and Moczko, J. (1990) Contribution of the soil heat flux to the heat balance of the active surface. *Zeszyty Akademii Rolniczej.* Kraków. **246**, 117-129.
Kedziora, A. (1995) *Podstawy agrometeorologii.* Panstwowe Wydawnictwo Rolnicze i Lesne, Poznan, pp 264.
Kedziora, A., Olejnik, J. and Kapuscinski, J. (1989) Impact of landscape structure on heat and water balance. *Ecology International,* **17**, 1-17.
Kedziora, A., Olejnik, J., Moczko, J. and Kazimierczak, R. (1987) Thermal and moisture conditions of a meadow and winter rape field. *Zeszyty Problemowe Postepów Nauk Rolniczych,* **322**, 105-135.
Knauer, N. and Mander, U. (1989) Untersuchungen über die Filterwirkung verschiedner Saumbiotope an Gewassern in Schleswig-Hilstein. *Z. F. Kulturtechnick und Landentwicklung,* **30**, 365-376.
Lee, G.F., Bentley, E. and Amundson, R. (1975) Effects of marshes on water quality. In: Hasler, A.D. (Ed.) *Coupling of land and water systems.* Springer Verlag, New York. pp 105-127.
Lowrance, R.R., Rodd, R.L. and Asmussen, L.E. (1983) Waterborne nutrient budgets for the riparian zone of an agricultural watershed. *Agriculture, Ecosystems and Environments,* **10**, 371-384.
Lowrance, R., Leonard, R. and Sheridan, J. (1985) Managing riparian ecosystems to control nonpoint pollution. *Journal of Soil and Water Conservation*, **40**, 87-91.
Marcinek, J., Spychalski, M. and Komisarek, J. (1990) Water cycling in an agricultural micro watershed (in Polish). In: Ryszkowski, L. and Kedziora, A. (Eds.) *Obieg wody i bariery biogeochemiczne w krajobrazie rolniczym.* Wydawnictwo Naukowe UAM Poznan, 69-96.
Molga, M. (1986) *Meteorologia rolnicza* (Agricultural meteorology). Panstwowe Wywadnictwo Rolinicze i Lesne, Warszawa. pp 492.
Muscutt, A.D., Harris, G.L., Bailey, S.W. and Davies, D.B. (1993) Buffer zones to improve water quality: a review of their potential use in UK agriculture. *Agriculture, Ecosystems and Environment,* **45**, 59-77.
Olejnik, J. and Kedziora, A. (1991) A model for heat and water balance estimation and its application to land use and climate variation. *Earth Surface Processes and Landforms,* **16**, 601-617.
Pauliukevicius, G. (1981) *Ecological role of the forest stand on the lake slopes.* Pergale, Vilnius. pp 191.
Peterjohn, W.T. and Correll, D.L. (1984) Nutrient dynamics in an agricultural watershed: observations on the role of a riparian forest. *Ecology,* **65**, 1466-1475.
Pinay, G. and Decamps, H. (1988) The role of riparian woods in regulating nitrogen fluxes between the alluvial aquifer and surface water: a conceptual model. *Regulated Rivers*, **2**, 507-516.
Prusinkiewicz, Z., Józefkowicz-Kotlarz, J., Kwiatkowska, A. and Pokojska, U. (1990) *The effect of shelterbelts on nutrient cycling in agricultural landscapes.* Poda a produkcia agroekosystemov. Bratislava. pp 108-126.
Ryszkowski, L. (1994) Strategy for increasing countryside resistance to environmental threats. In: Ryszkowski, L. and Balazy, S. (Eds.) *Functional appraisal of agricultural landscape in Europe.* Research Center for Agricultural and Forest Environment. Poznan. pp 9-18.
Ryszkowski, L., Bartoszewicz, A. (1987) Impact of agricultural landscape structure on cycling of inorganic nutrients. In: Clarholm, M. ans Bergström, L. (Eds.) *Ecology of arable land.* Kluwer Academic Publishers. pp 241-246.
Ryszkowski, L., Karg, J., Szpakowska, B. and Zyczynska-Baloniak, I. (1989) Distribution of phosphorus in meadow and cultivated field ecosystems. In: Tissen, H. (Ed.) *Phosphorus Cycles.* Turner-Warwick Communication, Saskatoon, Canada. pp 178-192.
Ryszkowski, L. and Kedziora, A. (1987) Impact of agricultural landscape structure on energy flow and water cycling. *Landscape Ecology,* **1**, 85-94.

Ryszkowski, L. and Kedziora, A. (1993) Energy control of matter fluxes through land-water ecotones in an agricultural landscape. *Hydrobiologia,* **251**, 239-248.
Soil Survey Staff (1975) *Soil Taxonomy: a basic system of soil classification for making and interpreting soil surveys.* Agric. Handbook 436. Washington D.C. pp 754.
Vanek, V. (1991) Riparian zones as a source of phosphorus for a groundwater-dominated lake. *Water Resources,* **25**, 409-418.
Wos, A. (1987) Mezoclimate of the environs of Turew. *Zeszyty Problemowe* Postepów Nauk Rolniczych, **322**, 13-82.

19 Modelling Internal Processes of Riparian Buffer Zones

A.J. Gold and D.Q. Kellogg

Department of Natural Resources Science,
University of Rhode Island, Kingston, RI 02881, USA.

"Chaos is the law of nature; order is the dream of man." Henry Adams (1918)

Abstract

We often develop models of the internal processes of buffer zones to predict the water-quality abatement potential associated with buffer zones of varying width, location and site characteristics. We also create models to illuminate our understanding of the interactions of water-quality contaminants with the biotic and abiotic factors within a buffer zone. In this paper, we explore a variety of models that simulate the fate of water-quality contaminants in buffer zones. Because of the central nature of hydrology to water-quality functions of buffer zones, we focus on the hydrologic components of buffer-zone models with special attention to the site-specific conditions that affect the flow path, residence time and soil environment encountered by water quality contaminants within a buffer zone. Throughout the paper, we address the effects of varying levels of spatial resolution on the ability of models to capture the site-specific processes inherent in buffer zones.

Application of the appropriate removal processes to waterborne contaminants requires estimates of flow paths and retention times within different portions of a buffer zone. Models can simulate the potential for contaminants to infiltrate into the soil or enter the buffer zone by subsurface flow. Once within the soil, methods exist to simulate a host of removal mechanisms including filtration, microbial transformations, plant uptake and adsorption. Modellers have also addressed situations where waterborne contaminants within the soil bypass rootzone process. For example, contaminated groundwater from the upland may travel below the biologically active portions of the soil or it may emerge at surface seeps and travel overland across the buffer zone. In addition, where preferential and macropore flow is likely, models can be constructed to reduce the expected groundwater residence time and related transformations.

Considerable information on buffer-zone processes can be inferred from conceptual or qualitative models that rely on spatial data, such as hydrogeomorphoric setting, vegetation type, slope and the soil classification. Site-specific soil data such as texture, structure and depth to redoximorphic features will refine these models. Through the use of pedotransfer functions, these same data can serve to initialize more quantitative models that require information on soil moisture characteristic curves and hydraulic conductivity. Clearly, more mechanistic modelling approaches should require more explicit spatial and temporal inputs. In this paper, we review various methods used to simulate infiltration, the extent of saturation within the rootzone and groundwater seepage. In theory, to simulate these processes requires linkage to hillslope hydrologic factors – such as incoming flow rates and the relative elevation of the downgradient receiving water. These external hydrologic characteristics are highly dynamic, and models exist that incorporate the effects of daily fluctuations of external factors. However, given the seasonality of overland runoff and groundwater flow, useful insights may be obtained from static simulations representing the expected conditions within hydrologically active periods of the year.

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0

INTRODUCTION

Vegetated buffer zones hold great promise for protecting streams and ponds from contaminated runoff and groundwater leaving upland areas. Because the performance of buffers is somewhat site-specific, there is great interest in new approaches to evaluate the water-quality functions of vegetated buffers. In this chapter, we describe models that focus on internal processes in buffers.

Why model internal processes? We recognize that field work is essential to any assessment of the water-quality functions of buffer zones. However, models provide us with the unique opportunity to study combinations of conditions which have not yet been encountered in field studies. With models, we can step back and gain a broader view with the purpose of detecting common patterns among sets of climatic and landscape conditions. This type of exercise would be difficult if we restricted ourselves to field data alone. Models of internal processes in riparian zones can also augment the development of qualitative management models. For example, models could be employed to evaluate and modify classification schemes that rank the potential capability of riparian zones to perform various water quality functions based on combinations of landscape, geology and land cover (Pinay and Décamps, 1988; Lowrance, 1997).

Models of water-quality functions in buffer zones encompass a wide range of approaches and complexity, arising mainly from the variety of the intended use and audience of each model. Addiscott (1993, 1994) and Burrough (1995) have thoughtfully suggested that the nature and intent of simulation models within the earth and ecosystem sciences ranges between two primary goals:

1) to develop decision support tools for resource managers
2) to increase our understanding of the interactions and mechanisms that affect processes within the landscape.

Addiscott (1994) terms decision support models as "functional" models, in contrast to process level "mechanistic" models. Typically, functional models require less explicit data inputs and rely on conceptualizations or abstractions to simplify the complex mix of physical and biological processes that exist in nature. Functional models often incorporate some mechanistic approaches, thus the distinction between the categories is often based on the intent rather than the exact algorithms within the models.

We will describe a number of mechanistic approaches to buffer zone modelling that can be used to examine and quantify site-specific processes at daily time scales. These approaches are linked by their requirements for intensive and often expensive data collection. To capture the functions and processes within a buffer zone, mechanistic models can be applied and compared to selected intensively monitored reference buffer zones to permit evaluation of the appropriateness of our process level understanding to real-life situations. Ideally, reference buffer zones would be located in a variety of climatic and hydrogeomorphic settings (Brinson, 1995). We suggest that mechanistic approaches are needed to establish the governing principles surrounding buffer zone processes. Our challenge is to find the lessons within the mechanistic models, honouring those insights while creating modelling approaches that recognize the limits imposed by the paucity of site-specific data.

How do we assess the accuracy of our models? Great controversy surrounds the issue of validation of landscape scale models (Addiscott, 1993). Bredehoeft and Konikow (1993) have suggested that we term our evaluation procedures "history matching", rather than validation. In this fashion we acknowledge that our model has agreed with measurements from a fixed set of situations and may not be suitable for all purposes. The evaluation of process-level models is confounded by the fact that many properties have high spatial and temporal variation. We will describe methods that incorporate variability of model inputs, and permit a "risk assessment" approach to the model outputs.

PHYSICAL SETTING

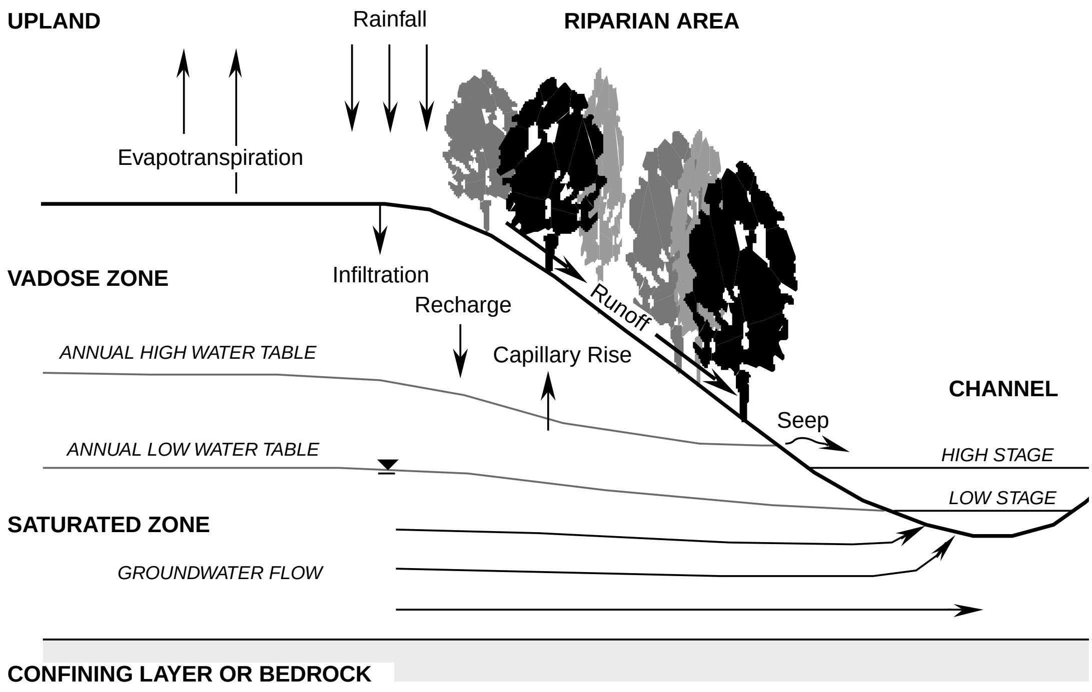
Our focus is on models that describe buffer zones located below upland areas and upgradient of open water bodies, such as streams, rivers, lakes and estuaries. These areas are typically referred to as "riparian" zones, from the Latin word ripa, meaning bank or shoreline. In humid regions riparian areas typically receive groundwater and surface water from upland areas. The riparian buffer zone can influence the quantity, timing and quality of those waters prior to entry into concentrated flow channels or open water.

The riparian ecosystem extends laterally from an amorphous upland boundary to the open water. There are many ongoing investigations that are attempting to define the upland boundary of riparian zone functions as well as the soil depth at which transformation rates compare to those found in deep groundwater. Generally, investigators believe the unique properties of riparian zones are found within the upper 1-2 m of the soil (hereafter known as the biologically active zone) and extend upgradient to the limit of soils that show evidence of periods of partial saturation.

Many riparian zones occur on sites with shallow water tables; locations where the water table is within 0.1 – 1.5 m from the soil surface. The water table is the top of the groundwater, or in the language of the physicist (Hillel, 1980), the location at which the soil water is at atmospheric pressure. Above the water table the soil water is under tension from capillary suction, while below the water table the water is subject to hydrostatic pressures. The depth of the water table in relationship to the biologically active zone has extraordinary implications for the internal processes within a riparian buffer zone.

Waterborne materials enter a riparian ecosystem via surface runoff, groundwater flow and precipitation, and leave the ecosystem through evapotranspiration, percolation to deep groundwater, surface seeps and flow to surface waters (Fig. 1). Our models need to reflect the processes that influence the flow path and residence times in different portions of the riparian ecosystem. For example, infiltration alters the flow path from surface transport to groundwater transport. As a result, the fluxes of sediment and sediment-bound contaminants through the buffer zone are greatly reduced. Conversely, groundwater that emerges in surface seeps will rapidly traverse a buffer zone and bypass most attenuation processes.

Figure 1. Schematic diagram of the hydrological cycle of a riparian buffer zone.



MODELS OF SOIL MOISTURE IN THE UNSATURATED ZONE

Elevated soil moisture is the hallmark of riparian buffer zones. Soil moisture can be expressed in several formats. On a volumetric basis:

$$\theta_v = V_w / V_t \qquad (1)$$

where:

θ_v = volumetric water content (cm^3/cm^3)

V_w = volume of water (cm^3)

V_t = total volume of the soil (cm^3)

and as % water-filled pore space:

$$WFP = \theta_v / \phi * 100 \qquad (2)$$

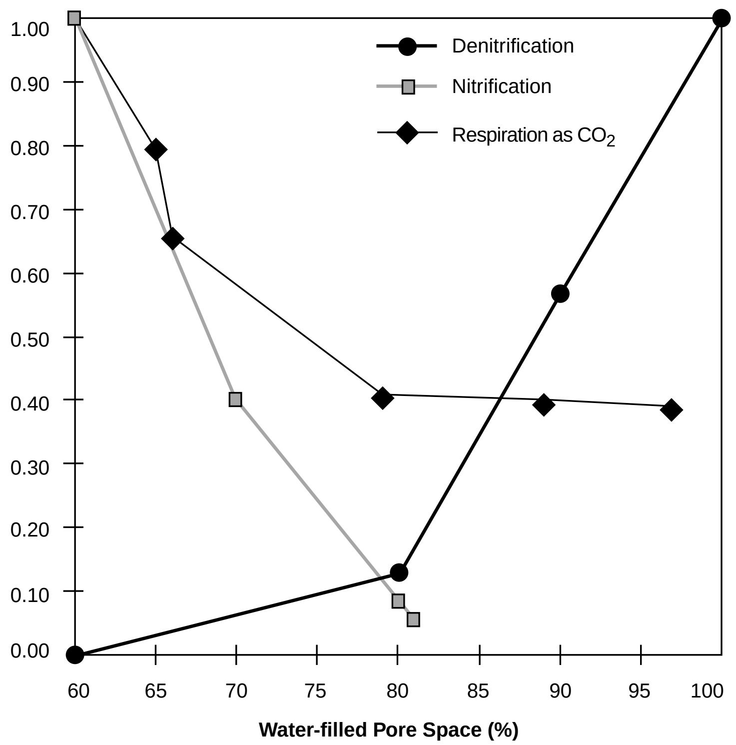
where:

WFP = % water-filled pore space

ϕ = total porosity (cm^3/cm^3).

Many internal processes within riparian ecosystems emanate from the temporal and spatial patterns of soil moisture within the biologically active zone of the soil. Linn and Doran (1984) related the relative potential for aerobic and anaerobic microbial activity to the WFP in a soil (Fig. 2). They concluded that the rate of many soil microbial activities, including nitrification, denitrification and respiration, is strongly influenced by WFP. In particular, relative microbial activity was extremely sensitive to WFP in the range between 60-100%. For example, in Figure 2, the relative denitrification rate within a soil increases 5-fold when the WFP increases from 80% to 90%.

Figure 2. Relationships of water-filled porosity versus selected microbial transformation rates. These relationships are commonly used in models of soil biogeochemistry. Adapted from Linn and Doran (1984).



Although Groffman and Tiedje (1988) have suggested that the relationships between WFP and microbial activity vary during wetting and drying cycles, several well-recognized soil biogeochemical models, such as the CENTURY model (Parton *et al.*, 1988) and the just-released REMM model (Altier *et al.*, in prep.) rely on the insights and relationships of Linn and Doran (1984) to modify microbial transformation rates based on WFP.

In addition to microbial processes, soil wetness can influence other processes, including the redox potential, infiltration rate, root uptake, evapotranspiration and the extent and timing of water table fluctuations. Thus, the ability to predict temporal and spatial patterns of WFP is central to explicit models of internal processes in riparian buffer zones.

There are a large number of one-dimensional soil-moisture models in use to evaluate the fate and transport of agrichemicals in upland areas. Efforts are underway to adapt these models for use in riparian areas. We will first review the capacity-based approach. This approach is the core of simulation models such as GLEAMS (Knisel *et al.*, 1992), SLIM (Addiscott and Whitmore, 1991), NLEAP (Shaffer *et al.*, 1991) and PRZM (Carsel *et al.*, 1984). In upland situations these functional models have been found to generate reasonable agreement with more mechanistic estimates of water and solute flux through the soil profile. However, in shallow water table soils, several of the assumptions inherent in capacity-based models are likely to generate misleading results. We will present a mechanistic model, LEACHN (Hutson and Wagenet, 1992), that can address these complex soil water dynamics. Finally, we will highlight where capacity-based models need to be modified to address riparian situations and present a hybrid model DRAINMOD (Skaggs, 1991), that may serve as a starting point for future work.

Capacity-based models

Capacity-based models divide the soil profile into a series of horizontal layers or compartments and generate a daily water balance for each layer. The mass balance equation for each soil layer is:

$$SM_{ij} = F_{ij} + D_{ij} - ET_{ij} - O_{ij} + SM_{i(j-1)} \qquad (3)$$

where:

SM_{ij} = soil moisture within layer i, on day j (cm)
F_{ij} = infiltration to layer i, on day j (cm)
D_{ij} = percolation received from drainage of upper soil layers on day j (cm)
ET_{ij} = plant and soil evapotranspiration from layer i, on day j (cm)
O_{ij} = drainage outflow from layer i, on day j (cm).

The volumetric water content (θ_v) of layer i, on day j is then computed as:

$$\theta_v = SM_{ij} \ / \ (\text{Vertical depth of soil layer i}) \qquad (4)$$

Capacity models ascribe three tiers of water retention capacities to each soil layer. The maximum water storage is the saturation water content, equivalent to 100% WFP. Wet soil is expected to drain to "field capacity", an equilibrium condition that occurs following drainage of gravitational water. In most capacity models, field capacity is fixed through time at a moisture content that equates to the soil water retention at –0.33 bars potential. Finally, each soil layer is assigned a capacity that corresponds to the water content unavailable for plant uptake, known as "permanent wilting point". Given the importance of partial saturation to microbial activity rates, both the rate of drainage from a soil layer as well as the equilibrium water content following gravitational drainage require careful examination.

In early versions of capacity models, gravitational water (i.e., soil water content above field capacity) was assumed to drain from a layer in a single day. In order to mimic drainage patterns more realistically, Addiscott and Whitmore (1991) created a functional rate parameter, α, that limits drainage to a set proportion of gravitational water each day. Through the use of this type of functional rate parameter, capacity models may be capable of providing some important insights into the temporal patterns of partial saturation within the biologically active zones of riparian buffers.

Capacity models recognize that infiltration is affected by soil moisture. Most models of infiltration, whether functional or mechanistic, rely on estimates of soil wetness of the surface layers to modify the infiltration rate for a given site. Drier conditions enhance infiltration, while wetter conditions may sometimes preclude infiltration or even contribute to overland flows via surface seeps. Common approaches that may be useful for riparian buffer zones are the Green-Ampt method and the Curve Number method (Rawls *et al.*, 1993).

Limitations of capacity-based models for riparian areas

As mentioned earlier, several assumptions inherent in capacity-based models do not match the soil-water dynamics in soils with shallow water tables:

1) Field capacity is a dynamic property in riparian soils

In sites with shallow water tables the soil moisture at any depth within the root zone is intimately linked to the depth of the water table. During periods of minimal evapotranspiration and infiltration the distribution of soil moisture in the unsaturated zone reaches an equilibrium with the water table in response to its pore size distribution and its distance above the water table. At equilibrium, soil water at a given location is held by a capillary tension equal to its vertical distance above the water table i.e., the potential pull exerted by gravity. Neglecting hysteresis, for a given soil, each capillary tension relates to a distinct soil water content. This relationship is known as the Soil Moisture Retention Curve (Hendrickx, 1990).

Figs. 3a & b illustrate the effect of the water table on the equilibrium water content at a depth of 20 cm within the same soil. When the water table is 100 cm from the surface, the equilibrium water potential and WFP are –80 cm (–0.08 bars) and 59% respectively. If the water table rises to 40 cm from the surface, the equilibrium potential and corresponding WFP would be –20 cm and 88%, respectively. Thus the potential for anaerobiosis in the unsaturated zone is much greater when the water table is 40 cm from the surface than when it is 100 cm from the surface — even when all other factors are unchanged.

Figure 3a. Soil water distribution and water potential at equilibrium conditions for a water table at 100 cm.

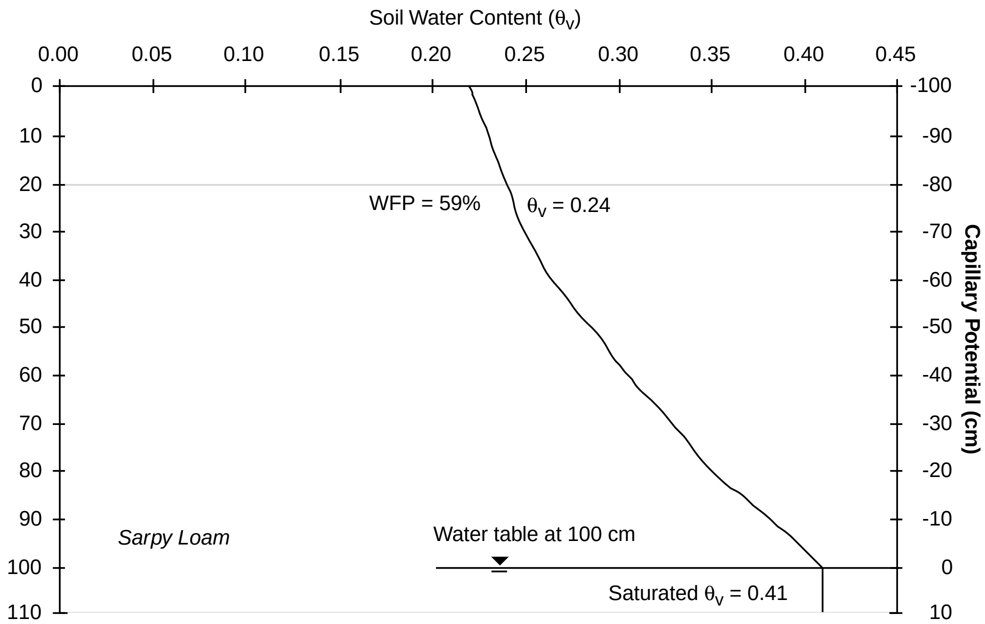
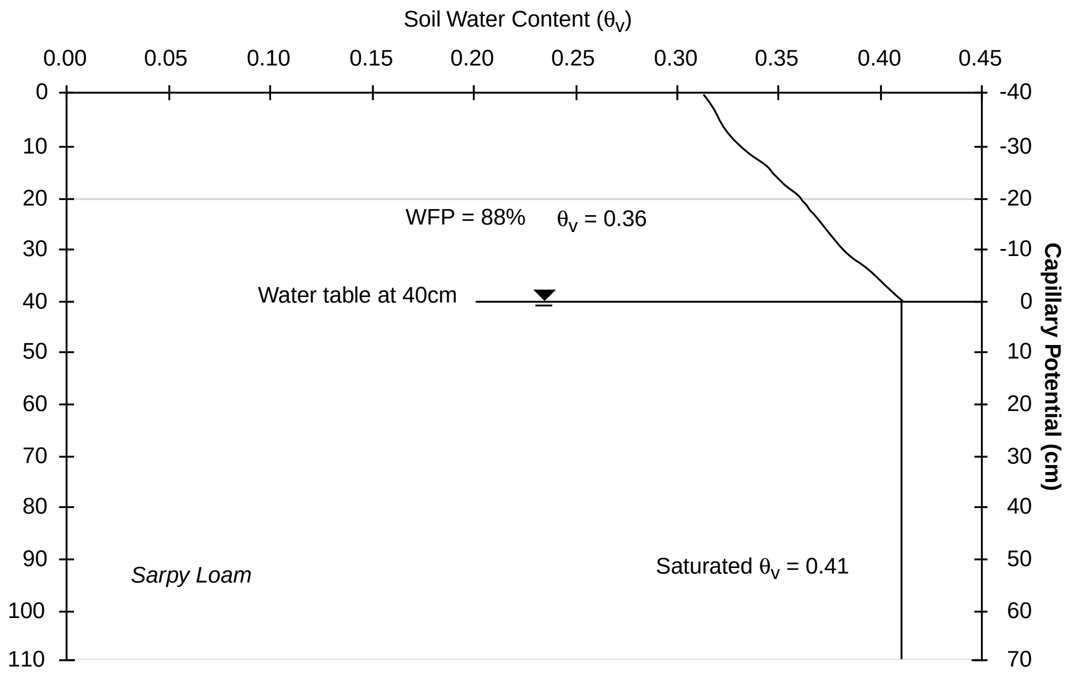


Figure 3b. Soil water distribution and water potential at equilibrium conditions for a water table at 40 cm.



2) Upwelling of water from the water table can occur in response to evapotranspiration

The presence of a shallow water table can affect the extent of water extraction and subsequent soil drying that results from evapotranspiration. Typical capacity-based soil moisture models incorporate a threshold soil moisture level to correspond to drought stress. Below this level, removal of soil moisture by evapotranspiration drops below non-stressed "potential" rates.

In upland situations, water extracted from the root zone is not expected to be replenished from greater depths and the soil moisture declines in direct response to evapotranspiration. However, in the presence of a shallow water table evapotranspiration can generate an upward flow of water to the root zone. The potential for upwelling results from a gradient between the drying soil within the root zone and the tension-free groundwater. The hydraulic conductivity of the media between the groundwater and the root zone is the dominant factor affecting the rate of upward flow. A version of the Darcy-Buckingham equation describes this upward flux:

$$q = K(\Psi)\,[(d(\Psi)/dz) - 1] \qquad (5)$$

where:

q = flux (equal to evaporation rate under steady-state conditions)

K(Ψ) = unsaturated hydraulic conductivity corresponding to a given soil moisture tension

Ψ□ = soil moisture tension

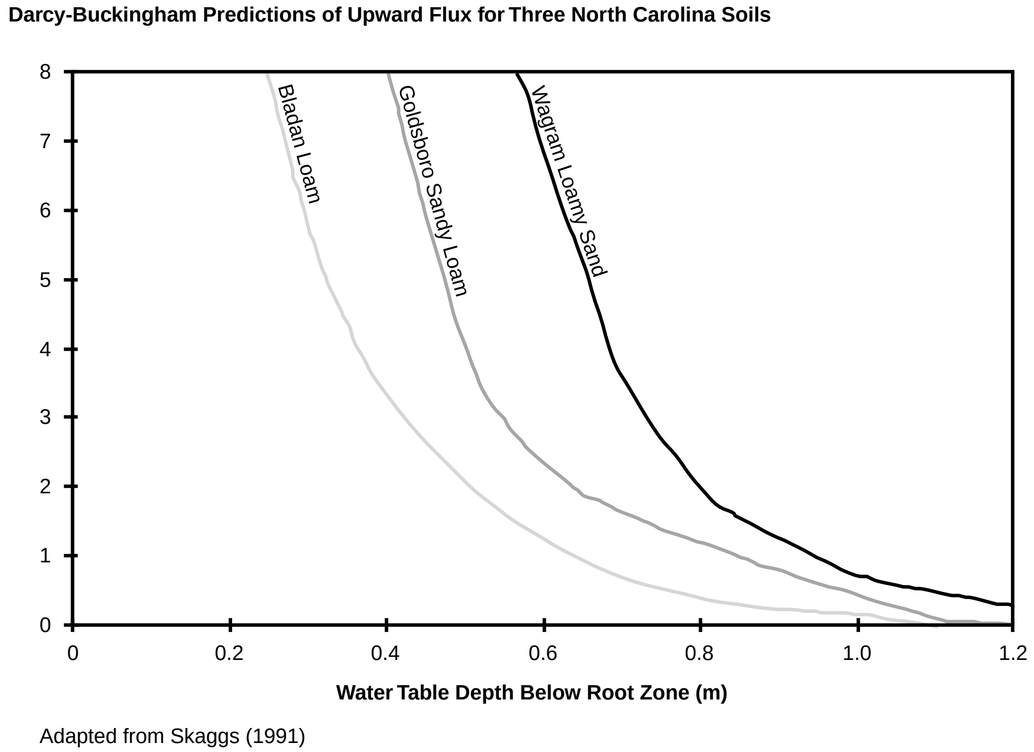
z = height above the water table.

The Darcy-Buckingham equation modifies Darcy's equation to address flux within unsaturated media. It reflects the fact that hydraulic conductivity for a given soil horizon is greatest when the soil is saturated and declines as soil moisture declines.

Skaggs (1991) used a mechanistic approach similar to that outlined in equation (5) to generate the relationships between depth to water and upward flux for different soils in North Carolina (Fig. 4).

These types of analyses have demonstrated that a wide range of soils can generate considerable upward flux in response to an evaporating zone within 1 m of the water table (Hillel, 1980; Skaggs, 1991).

Figure 4. Effect of water table depth on steady upward flux from the water table for several North Carolina soils. From Skaggs (1991).



The occurrence of upwelling into the root zone demands that we incorporate several factors into our models of riparian zone dynamics. First, we must recognize that groundwater contaminants such as nitrate-N have the potential to move into the biologically active zone of the unsaturated zone. In that setting, nitrate-N will be subject to a host of biological transformations, including plant uptake, denitrification and immobilization. Second, because of upwelling, the entire soil horizon of riparian sites can remain considerably wetter than upland soils. In the presence of a shallow water table, the root zone will begin to dry only if the rate of evapotranspiration exceeds the rate of upwelling into the root zone. As a result, we need to adjust the common capacity-based approaches to soil moisture balance simulations to reflect the fact that minimal drying may result from evapotranspiration.

Mechanistic models of soil moisture

Mechanistic models, e.g. LEACHN (Hutson and Wagenet, 1992), have been developed to simulate the movement of water and solutes through soil profiles. Some of these models use daily or hourly climatic records and permit evaluations of flux in response to seasonal and annual fluctuations. These models rely on the Darcy-Buckingham equation in combination with the continuity equation to generate soil moisture flux in the unsaturated zone. Models can simulate the full suite of conditions that occur in soil with shallow water tables. The key data requirements are soil hydraulic properties, particularly the soil moisture retention curve (θ vs. Ψ) and the relationship between hydraulic conductivity and volumetric water content (K vs. Ψ).

Several excellent reviews (Hendrickx, 1990; Rawls *et al.*, 1993; Mualem, 1986) describe the use of field, laboratory and statistical techniques to estimate soil hydraulic properties. The soil moisture retention curve is often generated in the laboratory with a variety of devices that induce a series of known pressures or suctions on undisturbed soil samples. The samples usually start at a saturated condition and are weighed after reaching equilibrium with each succeeding pressure or suction. Hanging water columns are used in combination with sand tables or pressure cells to simulate near saturated conditions. For greater tensions, some type of pressurized chamber is required. Occasionally, the soil moisture retention curve is obtained under field conditions. This approach can be extremely time-consuming and requires the use of non-destructive monitoring methods, such as time domain reflectometry in combination with tensiometers. Both laboratory and field methods are time and labour intensive.

The relationship between K and Ψ is even more difficult to obtain directly from field and laboratory measurements. As a result, hydraulic properties of soils are often generated from pedotransfer functions (Bouma and van Lanen, 1987; Bouma *et al.*, 1996). Pedotransfer functions relate soil hydraulic properties to more easily measured soil properties, such as texture or the saturated hydraulic conductivity. In general, all of these relationships are recommended for use with generic classes of soils, rather than for site-specific analyses. More reliable pedotransfer functions have also been developed which use selected field measurements of soil hydraulic properties to generate the full range of relationships between K and Ψ and between Ψ and θ (van Genuchten *et al.*, 1991).

An additional difficulty associated with the prediction of soil moisture in the unsaturated zone is that the soil moisture retention curve does not follow the same pattern during periods of drainage or wetting. This phenomenon, known as hysteresis, theoretically can have a large influence on the capillary fringe above the water table (Jaynes, 1990). Very limited information is available on the extent of hysteresis in soils and the incorporation of algorithms that describe hysteresis would add great complexity to soil water models.

WE NEED TO INCORPORATE VARIATION INTO OUR MODELS

We have argued that riparian buffer zone models need to be highly sensitive to the location of the water table and to soil hydraulic properties in the unsaturated zone. Yet, both of these characteristics can display a high degree of spatial variability, particularly since riparian areas are transition areas between uplands and open water. In addition to spatial variation, the water table depth and corresponding responses in the unsaturated zone will exhibit seasonal trends that are confounded by intermittent recharge events (Haycock and Pinay, 1993).

The soil moisture models we have described simulate processes at a point scale, based on assumptions of Darcian flow. These types of models have come under well-deserved criticism for their failure to describe the range and complexities of landscape-scale processes. Of additional note is the fact that we often aggregate selected areas of land into classes of similar characteristics and select a single value to represent the input parameters of the aggregated area (Beven, 1989). However, within a soil mapping unit the soil hydraulic properties are notoriously variable at the point scale (Warrick and Nielson, 1980). Soil survey maps are the most available source of input data for buffer zone modellers. Yet, Simmons *et al.* (1992) found large variation in the water table elevation within a single soil drainage class aggregate of a soil survey map, suggesting that the resolution of standard soil maps (13 m at a scale of 1:15,840) may be too coarse to incorporate important differences between soil classes.

Hence, any deterministic approach may create a misleading picture of the magnitude and relative importance of different riparian processes. Equally important, a deterministic approach ignores the need to quantify the level of uncertainty inherent in our simulations. We are likely to find that different types of buffer areas will vary in both the expected mean as well as in the variability of their water quality functions. Both of these factors will be of great use to decision makers who need to consider risk assessment in their management strategies (Bouma *et al.*, 1996).

Armed with the knowledge that critical inputs have a high degree of spatial and temporal variability, we should strive to build some level of stochastic evaluation into our modelling efforts. Monte Carlo techniques offer a useful method to incorporate uncertainty into deterministic models (Ross, 1990). In the Monte Carlo method selected input variables are represented by a probability distribution function (PDF), rather than by a single value. A series of simulations, sometimes referred to as an 'ensemble', is then executed. In each simulation the input is obtained from a random selection of values based on the PDF of each variable. Each simulation then represents an equally likely realisation. Thus, the mean, standard deviation, and other statistical properties of the predicted outputs are obtained.

Because of the transitional location of riparian buffer zones, certain riparian zones will need to be divided into a number of unique settings to represent individual drainage classes or landscape units that differ in such characteristics as root zone depth, soil texture, organic matter, wetness and soil hydraulic properties. In these types of situations a Monte Carlo approach would require many iterations of a model. Even with the recent improvements in computation speed, Monte Carlo simulations can demand enormous computational time for mechanistic models. A simple sensitivity analysis, performed to highlight the effects of varying any single parameter on model outcome, does not generate a sense of the interaction of the input parameters on the expected outcomes. Addiscott and Wagenet (1985) developed a simplified technique to investigate simultaneously the effects of variability from multiple parameters. This approach requires far fewer iterations than Monte Carlo techniques and still provides insight into the likely variation of the simulated outcomes.

A POSSIBLE SOLUTION; A HYBRID APPROACH TO MODEL RIPARIAN PROCESSES

Capacity-based soil moisture models are relatively simple to program and can rapidly assess the effects of temporal and spatial variability at a site. With several alterations these models could reflect the unique aspects of riparian zones. For example, the dynamics of the unsaturated zone in the DRAINMOD water management model (Skaggs, 1991) are simulated through a modification of capacity-based models. This model predicts the hourly response of the water table and the unsaturated zone to hydrologic factors. Like capacity-based models, a hydrologic balance is computed at each time interval for the unsaturated zone. However, rather than relying on a fixed value for field capacity, the aeration porosity that equates with the "field capacity" of the unsaturated zone is recalculated on an hourly basis. The model assumes that the soil above the water table will reach the equilibrium soil wetness predicted from its soil moisture retention curve and its vertical distance to the water table. Then, in a similar fashion to capacity-based models, gravitational drainage from the unsaturated zone occurs when the moisture content of the layer is above the newly calculated field capacity.

The potential extent of gravitational drainage associated with varying water table depths is a required input to the model. Thus, the computing time needed to accommodate drainage is minimal compared to the intensive iterations required by a mechanistic model. DRAINMOD also incorporates the relationship between the depth to the water table and the maximum upward flux in response to evapotranspiration (Fig. 4). Again, the model does not develop this relationship, rather, it is an input requirement. These "look-up" tables can be generated separately through a variety of modelling approaches. By eliminating these computations from the actual model, the physics and mathematics in the model are relatively straightforward. DRAINMOD predictions of the effects of a shallow water table on soil moisture in the unsaturated zone have compared favorably with the results obtained with a mechanistic model.

The DRAINMOD model is not capable of simulating water table fluctuations in natural riparian zones. DRAINMOD uses well-established methods to estimate the response of the water table in situations with artificial drainage. The boundary conditions are well-defined, and the simulations that predict the spatial and temporal patterns in water table elevations are controlled by relationships based on fixed spacings between parallel drainage tubes or ditches. In riparian zones, the boundary conditions can vary widely, increasing the complexity of simulations of water table depths.

WATER TABLE DYNAMICS MUST BE RECOGNIZED IN RIPARIAN AREAS

The depth to the water table in riparian areas is determined by temporally and spatially variable conditions at the defined boundaries and by the law of mass conservation. Figure 1 shows groundwater flow through a riparian area in two dimensions. The water table elevation depends on a mix of hydrology, geomorphic setting and soil properties, and can exhibit substantial spatial and temporal variability at any one site. During the course of the year, the water table usually rises in the dormant season and falls during the growing season. Bosch *et al.* (1994) demonstrated a marked seasonal difference in groundwater flow patterns through a riparian area due to changes in evapotranspiration demands and input from uplands.

Flow is bounded in the x-direction by input from upland (upgradient) saturated flow, and by the elevation of the downgradient stream to which groundwater is flowing. Vertically (i.e. in the z-direction), the system behaves essentially like an unconfined aquifer, where flow is constrained by a confining layer or bedrock below. Above, there can be input from infiltrating percolate, as well as output via (a) a vertical gradient created by evapotranspiration demands, and (b) surface seeps where shallow flow paths intersect the soil surface. The water balance of the saturated zone can be represented by the equation:

$$Q_{upland} + Q_{infil} = Q_{stream} + Q_{et} + Q_{seep} + DQ_{store} \qquad (6)$$

where:

Q_{upland} = groundwater inflow from the upland (m^3/day)

Q_{infil} = contribution to groundwater from infiltrating percolate

Q_{stream} = groundwater flow to the stream

Q_{et} = movement of water to the soil surface and plant roots as a result of evapotranspiration

Q_{seep} = groundwater seepage to the soil surface due to the inability of the soil profile to carry the total flow in the subsurface

DQ_{store} = change in groundwater storage.

A common approach to modelling shallow groundwater flow towards a stream or drain is accomplished by applying what are known as the Dupuit-Forchheimer assumptions (Hillel, 1977; Skaggs, 1991), where it is assumed that all saturated flow is horizontal and that the rate of flow is proportional to the slope of the water table. These assumptions are best applied to situations where the horizontal extent of the flow is large relative to the soil depth, and should be valid for many riparian zone landscapes. Flow rate is characterized by:

$$q = K_s iH \qquad (7)$$

where:

q = rate of groundwater flow through a unit thickness of soil (m^3/day/m)

Ks = saturated hydraulic conductivity (m/day)

i = hydraulic gradient, defined as the change in the hydraulic head over a given distance (m/m)

H = elevation of the water table above the confining layer (m).

The saturated hydraulic conductivity, K_s, is site-specific and dependent upon soil and fluid properties. K_s will remain unchanged over time, while i and H, the hydraulic gradient and the water table elevation respectively, will change in response to daily variations in storage (as a result of daily changes in inflows and outflows).

The extent of transformations of groundwater contaminants is a function of the transformation rates that occur in different locations along the flow path and the residence time of the groundwater within each location. The time of travel through a riparian area, also called the residence time, is inversely proportional to the saturated hydraulic conductivity:

$$t_r = L\phi/K_s i \tag{8}$$

where:

t_r = time a volume of water spends moving through a uniform section (day)

L = the straight line length of the path traversed through the soil (m)

ϕ = the porosity of the media (cm^3/cm^3).

If substantial flow through the biologically active zone occurs, a longer residence time allows for a higher degree of contaminant transformation.

However, the pattern of groundwater flow through a riparian buffer is dependent partially on the distribution of K_s throughout the soil profile. Layers with higher K_s will carry a greater portion of the flow. If the K_s of the biologically active zone is markedly less than that of the lower layers, much of the groundwater flow may follow deeper flow paths. Thus the groundwater can bypass the biologically active zone and enter a stream relatively unaffected by the presence of the riparian buffer.

The flow path, and the depth to the water table from the soil surface, are not only dependent upon the soil properties, but also on the boundary conditions, i.e. the inputs to and outputs from the system, as represented in equation (6). These in turn influence both the change in hydraulic head across a riparian area as well as the rate of groundwater flow through a riparian area. In particular, the downgradient stream elevation and the groundwater inflow from the upland are both intimately linked to the hydraulic gradient.

Boundary conditions and the depth of the soil profile affect the possibility for groundwater to emerge in surface seepage ($Q_{seep} > 0$). Seeps can generate channelized flow and effectively create a short-circuit to the stream, bypassing the biologically active zone (Warwick and Hill, 1988). Seeps occur where groundwater flow paths intersect the soil surface, such as at the bottom of hillslopes, or in areas where the soil is too shallow and K_s is too low to accommodate the total change in storage. The potential for seeps to occur can vary seasonally. For example, during the dormant season, inflow from the upland tends to be greater and the stream elevation higher, accompanied by higher precipitation and lower evapotranspiration. Where K_s is too low to accommodate this level of flow, seepage of excess groundwater flow to the soil surface will occur.

MODELS OF WATER TABLE DYNAMICS IN RIPARIAN AREAS

Methods for characterizing groundwater flow through a riparian area range from mechanistic models which update the water table using an hourly time step, to generalized functional models which characterize groundwater flow based on such factors as landscape, geology and soil redoximorphic features.

Mechanistic models to determine depth to water table

Several mechanistic models exist (e.g., MODFLOW), which employ either finite-element or finite-difference algorithms to describe subsurface hydrology at extremely small intervals of both space and time. These models rely on establishing boundary conditions and detailed information on spatial patterns of hydraulic conductivity, flow and/or hydraulic head (Wang and Anderson, 1982; Committee on Groundwater Modeling Assessment, 1990). Obviously, these models require extensive input data, which is variable and often difficult to obtain. The process of calibration and validation, however, is extremely valuable in that it demonstrates possible areas where further study is warranted or where more data collection is needed. This is the unqualified strength of mechanistic models and is the basis for worthwhile research. A hybrid approach, which combines mechanistic modelling with some generalizations, appears to be a good compromise for other kinds of investigations.

Hybrid models to determine depth to water table

Hillel (1977) devised a two-dimensional scheme to model water table dynamics in a hillslope. The soil profile is divided laterally into vertical columns and the columns are divided into layers, so that non-uniform properties can be specified both vertically and horizontally. Equations (6) and (7) are then applied, in each layer of each column, with $q = K_s \, i \, z$, where z is the layer thickness. The water table elevation is calculated on a daily basis, and can rise and fall in response to changes in q, based on the soil moisture retention curve. The upgradient boundary condition is defined as a watershed divide ($q_{upland} = 0$), while the downgradient boundary condition is specified as a constant stream elevation. The stream elevation could be specified on a daily basis, if it is known. For analysis of a riparian area, the water table algorithm could be modified to allow for groundwater inflow from the upland, rather than requiring the upgradient boundary be a watershed divide. However, additional input data would then be required to estimate groundwater flow into the riparian area from the upland, which is extremely difficult, if not impossible, to determine on a daily basis.

The most detailed and data-intensive analysis using this kind of hybrid approach would be a continuous daily time step over a period of several months or years. Collection of the necessary input data is time-consuming and expensive. In addition, calibration and validation for such a model are inherently unreliable due to the uncertainties and variability associated with the input data.

A simpler approach would be to evaluate the effects of different steady-state boundary conditions on the water table depth and unsaturated zone in the dormant and growing seasons. For example, a series of simulations could be performed with various combinations of q_{upland}, and stream elevations, along with a range of meteorological conditions. Seasonal differences could be honoured by matching the range of boundary conditions and inputs with those expected during the growing season or the dormant season. The meteorological input data could also be provided within a set range representing dry, average and wet years. This approach can yield valuable information about the expected range of conditions under which seeps occur, or the conditions which favor groundwater flow through the shallow biologically active zone.

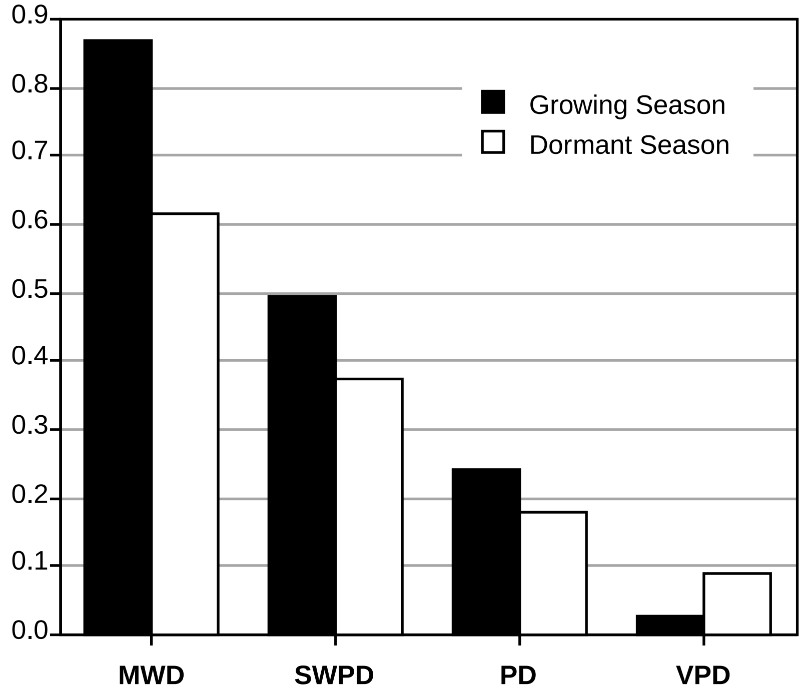
Functional approaches to water table estimates in riparian areas

Simulations of the spatial and temporal patterns of water table depths may be too unwieldy for most riparian models. Instead, we believe that the pattern of water table depth might be an input to the model. Again, a range of water table patterns should be used to reflect seasonal differences as well as the inherent uncertainty in this property. This may be the most practical approach if the purpose of the exercise is to learn enough about the system to be able to provide insights and/or recommendations to resource managers regarding the capability of the riparian area to attenuate different types of pollutants.

Information on the depth and pattern of the water table can be obtained through direct measurements or through soil morphological investigations. Water table wells are easy to install and to monitor. Site visits during periods of high water movement can also identify seepage faces and elucidate potential flow paths and retention times in different portions of the riparian area.

Considerable information about the depth and influence of the water table can be obtained from the drainage class of a soil. In the USA, soil series are classified into one of seven drainage classes based on the depth and extent of redoximorphic features that reflect the frequency, duration and seasonal timing of saturation or partial saturation during soil formation (Brady, 1974). Drainage classes differ in a number of characteristics that can affect soil processes, including the depth to the seasonal high water table, vertical distribution and percentage of organic matter in the solum, and timing and location of anaerobic conditions within the soil profile. Routine monitoring of water table wells can provide useful relationships to the depth and properties of soil redoximorphic features, thereby permitting extension of the sampling effort to other sites (Fig. 5). A number of field and laboratory studies (Simmons *et al.*, 1992; Groffman *et al.*, 1992; Nelson *et al.*, 1995) have found drainage class to be a useful indicator of groundwater nitrate removal capacities within riparian areas.

Figure 5. Mean seasonal depth to the water table versus soil drainage class in a toposequence within a riparian buffer zone. Adpated from Simmons *et al.*, (1992).



CONCLUDING REMARKS

We have presented a variety of approaches to model internal processes that affect the water quality functions of riparian buffers. There remains considerable room for improvement. We are limited by a lack of data on the hydraulic properties of partially saturated soils. Standard pedotransfer functions need to be revised and targeted to riparian soils. Our data collection methods need to reflect the potential for "hot spots" to dominate the water quality functions of a landscape unit. Different sampling schemes and models may be required in situations where substantial infiltration occurs through preferential flow paths (Anderson and Burt, 1990) or where denitrification is primarily located in isolated patches (Parkin, 1987). Finally, we believe that the modeller can never lose sight of the field. Regardless of our ability to link together complex algorithms, personal observation of a site during periods of high water movement remains the key to our hopes to represent nature's complexity.

ACKNOWLEDGMENTS

The authors thank G.M. Cheschier for his helpful comments regarding the DRAINMOD model. Contribution no. 3411 from the Rhode Island Agricultural Experiment Station.

REFERENCES

Adams, H. (1918) *The education of Henry Adams.* Houghton Mifflin, Boston, MA. 519 pp.
Addiscott, T.M. (1993) Simulation modelling and soil behaviour. *Geoderma,* **60**, 15-40.
Addiscott, T.M. (1994) Simulation, prediction, foretelling or prophesy? Some thoughts on pedogenic modelling. In: Bryant, R.B. and Arnold, R.W. (Eds.) *Quantitative modeling of soil forming processes.* SSSA Special Publication 39. SSSA, Madison, WI. pp. 1-15.
Addiscott, T.M. and Wagenet, R.J. (1985) A simple method for combining soil properties that show variability. SSSAJ, **49**, 1365-1369.
Addiscott, T.M. and Whitmore, A.P. (1991) Simulation of solute leaching in soils of differing permeabilities. *Soil Use Manage.,* **7**, 94-102.
Altier, L.S., Lowrance, R., Williams, R.G., Sheridan, J.M., Hubbard, R.K., Bosch, D.D., Mills, C. and Thomas, D.L. (In preparation) *Riparian Ecosystem Management Model Documentation,* Version 1.0.
Anderson, M.G. and Burt, T.P. (1990) Subsurface runoff. In: Anderson, M.G. and Burt, T.P. (Eds.) *Process Studies in Hillslope Hydrology.* John Wiley & Sons, Chichester. 539 pp.
Beven, K. (1989) Changing ideas in hydrology; the case of physically-based models. *J. of Hydrology,* **105**, 157-172.
Bosch, D.D., Hubbard, R.K., West, L.T. and Lowrance, R.R. (1994) Subsurface flow patterns in a riparian buffer system. *Transactions of the ASAE,* Volume **37** (6), 1783-1790.
Bouma, J., Booltink, H.W.G. and Finke, P.A. (1996) Use of soil survey data for modeling solute transport in the vadose zone. *J. Environ. Qual.,* **25**, 519-526.
Bouma, J. and van Lanen, H.A.J. (1987) Transfer functions and threshold values: From soil characteristics to land qualities. In: *Quantified land evaluation.* Proc. Workshop ISSS/SSSA. ITC Publ., **6**, 106-111.
Brady, N. (1974) *The nature and properties of soils,* 8th ed. MacMillan Publ. Co., New York.
Bredehoeft, J.D. and Konikow, L.F. (1993) Ground-water models: Validate or invalidate. *Groundwater,* **31**, 178-179.
Brinson, M. (1995) The HGM approach explained. *National Wetlands Newsletter,* Nov-Dec. pp. 7-13.
Burrough, P.A. (1995) Opportunities and limitations of GIS-based modeling of solute transport at the regional scale. In: *Applications of GIS to the modeling of non-point source pollutants in the vadose zone.* Proceedings of 1995 Bouyoucos Conference, Riverside, CA, pp. 1-19.
Carsel, R.F., Smith, C.N., Mulkey, L.A., Dean, J.D. and Howise, P. (1984) *User's manual for the pesticide root zone model (PRZM),* release 1. USEPA-600/3-84-109. USEPA, Washington, DC.
Committee on Ground Water Modeling Assessment (1990) *Ground Water Models: Scientific and Regulatory Applications.* National Research Council. National Academy Press., Washington, DC. 303 pp.
Groffman, P.M., Gold, A.J. and Simmons, R.C. (1992) Nitrate dynamics in riparian forests: Microbial studies. *J. Environ. Qual.,* **21**, 666-671.
Groffman, P.M. and Tiedje, J.M. (1988) Denitrification hysteresis during wetting and drying cycles in soil. *SSSAJ,* **52**, 1626-1629.
Haycock, N.E. and Pinay, G. (1993) Groundwater nitrate dynamics in grass and poplar vegetated riparian buffer strips during the winter. *J. Environ. Qual.,* **22**, 273-278.
Hendrickx, J.M.H. (1990) Determination of hydraulic soil properties. In: Anderson, M.G. and Burt, T.P. (Eds.) *Process Studies in Hillslope Hydrology.* John Wiley & Sons, Chichester. 539 pp.
Hillel, D. (1977) *Computer Simulation of Soil-Water Dynamics.* International Development Research Centre, Ottawa, Canada.
Hillel, D. (1980) *Applications of Soil Physics.* Academic Press, New York. 385 pp.
Hutson, J.L. and Wagenet, R.J. (1992) LEACHM: *Leaching Estimation And Chemistry Model. Version 3.0.* Dep. Soil, Crop and Atmospheric Sciences. Res. Rep. no.92-3. Cornell University, Ithaca, NY.
Jaynes, D.B. (1990) Soil water hysteresis: Models and implications. In : Anderson, M.G. and Burt, T.P. (Eds.) *Process Studies in Hillslope Hydrology.* John Wiley & Sons, Chichester. 539 pp.
Knisel, W.D., Davis, F.M. and Leonard, R.A. (1992) *GLEAMS user manual. Version 2.0.* University of Georgia. Coastal Plain Experiment Station, Tifton, GA.
Linn, D.M. and Doran, J.W. (1984) Effect of water-filled pore space on carbon dioxide and nitrous oxide production in tilled and non-tilled soils. *SSSAJ,* **48**, 1267-1272.
Lowrance, R. (1997) The Potential Role of Riparian Forests as Buffer Zones. In: Haycock, N.E., Burt, T.P., Goulding, K.W.T. and Pinay, G. (Eds.) *Buffer Zones: Their Processes and Potential in Water Protection.,* Quest Environmental, UK.
Mualem, Y. (1986) Hydraulic conductivity of unsaturated soils: Prediction and formulas. In: Klute, A. (Ed.) *Methods of Soil Analysis.* Part 1. Monograph No. 9. Agronomy Series. ASA, SSSA, Madison, WI. 1188 pp.
Nelson, W.M., Gold, A.J. and Groffman, P.M. (1995) Spatial and temporal variation in groundwater nitrate removal in a riparian forest. *J. Environ. Qual.,* **24**, 691-699.
Parkin, T.B. (1987) Soil microsites as a source of denitrification variability. *SSSAJ,* **51**, 1194-1199.
Parton, W.J., Stewart, J.W.B. and Cole, C.V. (1988) Dynamics of C, N, P and S in grassland soils: a model. *Biogeochemistry,* **5**, 109-131.

Pinay, G. and Décamps, H. (1988) The role of riparian woods in regulating nitrogen fluxes between the alluvial aquifer and surface water: a conceptual model. *Regulated Rivers: Research and Management,* **Vol. 2**, 507-516.
Rawls, W.J., Ahuja, L.R., Brakensiek, D.L. and Sirmohammadi, A. (1993) Chapter Five. In: Maidment, D.M. (Ed.) *Handbook of Hydrology.* McGraw-Hill, Inc., New York, pp.5.1-5.51.
Ross, S.M. (1990) *A course in simulation.* MacMillan Publ. Co., New York.
Shaffer, M.J., Halvorson, A.D. and Pierce, F.J. (1991) Nitrate leaching and economic analysis package (NLEAP): Model description and application. In: Follet, R.F., Keeney, D.R. and Cruse, R.M. (Eds.) *Managing Nitrogen for Groundwater Quality and Farm Profitability.* SSSA, Madison, WI, pp. 285-322.
Simmons, R.C., Gold, A.J. and Groffman, P.M. (1992) Nitrate dynamics in riparian forests: Groundwater studies. *J. Environ. Qual.,* **21**, 656-665.
Skaggs, R.W. (1991) Drainage. In: *Modeling Plant and Soil Systems.* Agronomy Monograph No. 31. ASA-CSSA-SSSA, Madison, WI, pp. 205-243.
van Genuchten, M.Th., Leji, F.J. and Yates, F.R. (1991) The RETC codes for quantifying the hydraulic functions of unsaturated soils. U.S. Salinity Lab., Riverside, CA.
Wang, H.F. and Anderson, M.P. (1982) *Introduction to groundwater modeling.* W.H. Freeman and Co., San Francisco, CA. 237 pp.
Warrick, A.W. and Nielson, D.R. (1980) Spatial variability of soil physical properties in the field. In: Hillel, D. (Ed.) *Applications of Soil Physics.* Academic Press, New York, pp. 319-344.
Warwick, J. and Hill, A.R. (1988) Nitrate depletion in the riparian zone of a small woodland stream. *Hydrobiologia,* **157**, 231-240.

20 Modelling the interaction between buffer zones and the catchment

P. Merot and P. Durand

Institut National de la Recherche Agronomique (INRA), Unité de Science du Sol et de Bioclimatologie, 65 route de saint Brieuc, 35042 Rennes Cedex, France.

Abstract

The classical agricultural non-point source pollution models, such as ANSWERS or AGNPS, usually do not explicitly use the buffer zone concept, although their modular, or distributed, conception allows it in theory. In practice, the main obstacle is that hydrology and biogeochemistry are much more complex and less understood in buffer zones than in cultivated fields. Attempts to model this concept, usually in relation to the riparian area functioning, can be classified in two ways.

(1) Empirical models. Some descriptors of buffer zones are linked by stochastic relationships with biological or biogeochemical functions. For example, relationships have been established between the relative area of forested riparian zones and the streamwater chemical or biological quality; and between the hydrological regimes of the wetlands and their productivity. Furthermore, the seasonal or inter-annual variability of the stream discharge can be related to the functioning of the wetlands.

(2) Deterministic models. These are essentially hydrological models based on the concept of variable contributing area. These models are distributed or semi-distributed (e.g. based on distribution functions of spatial variables). Some of them are mechanistic models (e.g., IHDM), but the most widely used and developed, currently, are conceptual models of the TOPMODEL type. In this case, a simple description of the topographic control on the extension of the saturated area generally allows an adequate simulation of the hydrology of the saturated zone and of the catchment. Some attempts have been made to couple these models with water quality descriptions, but usually in a very crude way that does not actually describe the specific biogeochemistry of the saturated zone. The main reason for this is probably the important heterogeneity of this zone, in terms of soils, biogeochemistry and water pathways.

Other landscape structures that could act as buffer zones, such as hedges, have been very rarely considered in the models. Some studies have tried to describe the role of hedges in modifying the surface flow route and enhancing infiltration. A few models simulate the water cycle in hedges. The role of hedges as pollutant sinks is not yet modelled, and actually very little investigated.

The main conclusion of this review is that the interactions between the catchment and buffer zones have mostly been seen by modellers as the hydrological control of the catchment via the variable saturated area concept. They have not yet fully taken into account the control of water quality within a catchment by the different potential buffer zones.

INTRODUCTION

The hydrological regime or status of wetlands is recognised as a fundamental factor of ecological and geochemical processes (Badre *et al.*, poster at this conference), of primary productivity (Mitsch, 1988) or regulation of nutrient fluxes (Sanchez-Perez and Trémolières, poster at this conference). This status depends on the hydrological cycle in the catchment that includes the wetland. While most field studies describe the water condition of the wetlands, few of them draw attention to the hydrological framework and very few attempt to model the relationship between the catchment and the wetland.

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0

Mitsch (1988), for instance, reviewed coupled productivity-hydrology models of forested wetlands. He emphasised that the forested wetlands with a pulsing hydroperiod show greater net primary productivity than the continuously flooded ones. This leads to a relationship between the regime (stagnant, slowly flowing, seasonal flooding, drained) and the net productivity. He notes that “It remains to develop proper ecological models of forested wetlands to quantify these hydrological conditions, perhaps in terms of water depth, flooding duration and frequency, and water renewal rate (inflows/average water storage)”. Nevertheless the tentative model presented remains a model of internal processes of the wetland.

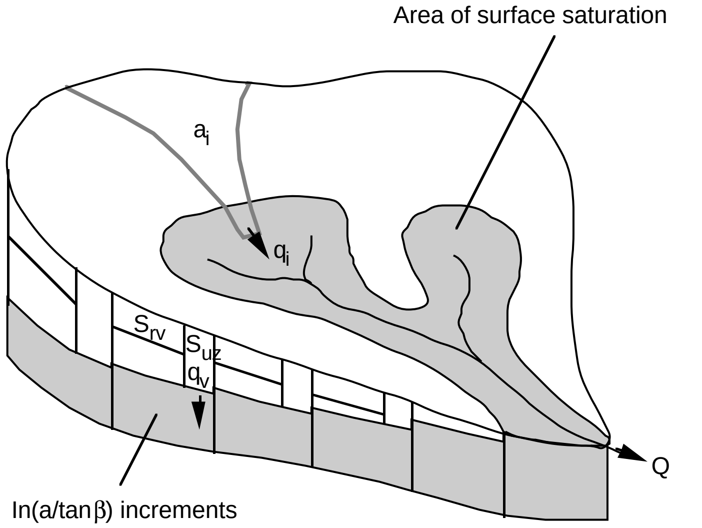
Distributed hydrological modelling seems the best way to answer Mitsch’s question. Therefore, we will present the way hydrologists take into account wetlands and how they model their behaviour within the catchment, emphasising the different implications for the spatial dynamics of wetlands and for flood generation processes. The major part of the paper will be devoted to wetlands as a buffer zone, but hedges can also act as buffer zones and will be considered briefly.

MODELLING THE HYDROLOGICAL INTERACTIONS BETWEEN THE VALLEY BOTTOM WETLAND AND THE CATCHMENT

A brief history

Many hydrologists consider that valley bottom wetlands play a key role in the water cycle, specifically in temperate climates. Early papers emphasising this role came from forest hydrologists who observed that in forested first-order catchments the infiltration capacity through soils was not limiting, because it was much greater than the rainfall intensity. One of the assumptions was that water flow during rainfall-runoff events came from the saturated areas of the catchment (Hewlett and Troendle, 1975; Ward, 1982), known as *variable source areas* (VSA). Cappus (1960), a French hydrologist, introduced a new theory of surface runoff, very close to the VSA concept: he considered that a small catchment may be divided in two parts of variable extension: 1) a zone of surface runoff due to the internal saturation of the soil, connected to the river, and 2) an infiltration zone, unsaturated and permeable, on slopes. The surface runoff coefficient at the outlet of the catchment can be approximately calculated by the relative area of the saturated zone.

Figure 1. Schematic representation of storage elements within a discrete ln(a/tan β) increment representation of a catchment area, showing the root zone store S_{rz}, vertical drainage store S_{uz} and recharge to the saturated zone, q_v, for one increment, and area, a_i, draining through a particular point i. The shaded area of surface saturation corresponding, in this case, to values of ln(a/tan β) falling in the highest class (adapted from Beven and Kirkby, 1979).



A lot of studies developed this concept of variable source area, or *contributing area.* Early studies showed the impact of the type of soil (Beven, 1978; Anderson and Burt, 1980), the importance of geomorphology on the dynamics of the extension with regards to the converging, diverging or parallel upslope flow lines (Anderson and Kneale, 1980; O'Loughlin, 1981) and the nature of water movement on and through the saturated areas (Kirkby, 1978).

A new hydrological modelling approach taking into account the concept of VSA can be considered as an important milestone in the consideration of the wetlands by hydrologists. In 1979, Beven and Kirkby presented a hydrological model "that attempts to combine the important effects of channel topology and dynamic contributing areas with the advantages of simple lumped parameter basin models". This variable contributing area model is presented in the next paragraph and in fig. 1.

The variable contributing area model (Beven and Kirkby, 1979)

The basic assumption of this model (TOPMODEL, from topography-based hydrological model), physically based on Darcy's law, is to consider that the hydraulic gradient on the slope is equal to the slope angle; the second assumption is that the hydraulic conductivity of the soil decreases exponentially with depth.

Under these assumptions, the discharge per unit width through the soil profile at each point i is given by:

$$q = K_0 \exp (S_i / m) \tan b.$$

where:

q = the flow downslope under an assumed hydraulic gradient due to gravity alone

K_0 = the saturated hydraulic conductivity at the soil surface

b = the local slope angle

S_i = the storage deficit (in rainfall equivalent unit)

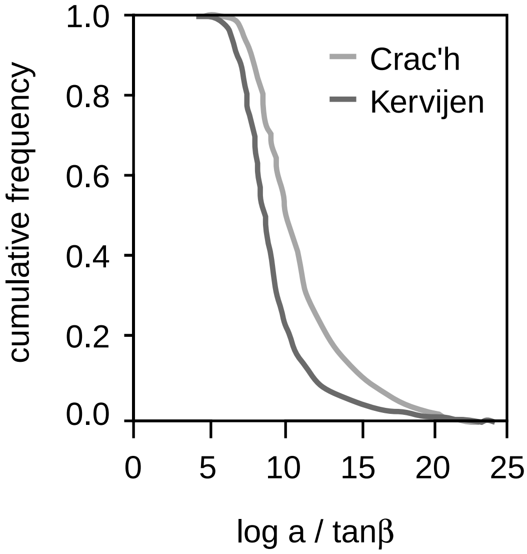
m = a constant

Developing this equation, the authors (Beven and Kirkby, 1979) show that the spatial distribution of the saturated areas in a catchment can be described by a topographic index, $\log (a/\tan \beta)$, where a is the drainage area per unit contour length. The potential saturation of soil increases with the value of this index. The frequency distribution of this index (Beven and Wood, 1983) can be regarded in fact as a synthetic representation of the sensitivity to saturation of different catchments (fig. 2). This first approach of modelling variable source areas was very successful and TOPMODEL is now a conceptual framework widely used and adapted. The most comprehensive presentation of TOPMODEL and its different applications appears in Beven *et al.* (1994).

The strength of this approach is twofold: this is the first model that takes into account in its construction the role of soil moisture distribution along a hillslope and the highly dynamic and non-linear behaviour of the saturated zone close to the hydrological network; secondly the model is based on elevation data which are very easy to obtain with the generalisation of Digital Terrain Models.

In the context of this volume, it is noteworthy that the contributing areas of hydrologists and the riparian zones of ecologists are two very similar entities. Therefore, it is quite surprising to note that the increasing interest of ecologists in wetlands over the last 20 years, and the discovery and enhancement of the variable source area concept by hydrologists have been parallel, but with very few connections until recently (Moore *et al.*, 1988).

Figure 2. Cumulative frequency of the topographic index for two catchments; one sensitive to saturation (Crac'h) and one less sensitive (Kervijen).



Development of TOPMODEL concept; complementary approaches

Moore *et al.* (1986) suggested a similar approach, based on a wetness index, where the saturation occurs when the drainage flow exceeds the transfer capacity of the soil profile. O'Loughlin (1981) developed the TOPOG model, based on the same concept of contributing area. Its interest relies on a more realistic description of the topography, based on contour lines and not on a grid scheme. Barling *et al.* (1994) proposed to replace the static topographic index by a quasi-dynamic wetness index. Ambroise *et al.* (1996) attempted to generalise the TOPMODEL concept, replacing the exponential decrease of conductivity by different functions related to the specific hydrological behaviour of catchments of interest.

Other modelling approaches

A lot of hydrological models or agricultural non-point source pollution models (Ag-NPSP), such as SHE (Abbot *et al.*, 1986), IHDM, ANSWERS, GLEAMS or AGPNS (Tim and Jolly, 1994), usually do not explicitly use the buffer zone concept, although their modular, or distributed conception allows it in theory. The SHE and IHDM models are mechanistic distributed hydrological models based on a regular or irregular grid description of the catchments. IHDM was designed for upland catchments with variable contributing areas, while SHE is more appropriate for regions with a strong regional component (Beven *et al.*, 1987). A version of the SHE adapted to contaminant transport- SHETRAN- is being developed (Ewen, 1995). Ag-NPSP models are usually built on field-scale models of sediment and chemical losses that are aggregated at the catchment scale. The limitations of such models for the assessment of buffer zone functioning are twofold. Firstly, hydrological and biogeochemical processes that are essential in the functioning of buffer zones have often been overlooked or only very crudely described in such models, which were implemented for other purposes. This is particularly the case for denitrification and lateral flow. Secondly, these models rely on the assumption that the parameters used for each individual cell are perfectly known and that the catchment response is the aggregation of the functioning of all the cells. In practice, it is impossible to have accurate information for all the parameters of each cell of the catchment and both the conception and the application of distributed models are fraught with acute scaling problems (Beven, 1993; Blöschl and Sivapalan, 1995).

In some large flat lands, the soil water content near the stream is related to groundwater dynamics and can be modelled by drainage formulae analogous to those used for agricultural drainage (De Vries, 1994). In an ancient salt marsh, currently devoted to farming, where 52% of the area had been drained for agricultural purposes, Giraud (1992) developed a comprehensive modelling approach coupling a drainage model for the fields (model SIDRA; Lesaffre, 1988) and a hydraulic model solving the St. Venant equations in a transient regime for a network of ditches and ponds (model MAGE). The quality of the simulation appears highly dependent on the topology of the network taken into account.

USING HYDROLOGICAL MODELLING TO PREDICT THE SPATIAL DISTRIBUTION OF VALLEY BOTTOM WETLANDS

While large wetlands are generally well-known due to their ecological interest, small wetlands scattered in the landscape, such as narrow strips along rivers, have seldom been studied. They tend to disappear, particularly in intensive farming zones, despite their considerable role as biogeochemical buffer zones. Therefore it is a major challenge to identify and locate these zones.

A first approach is to consider the soil characteristics. Wetlands soils exhibit particular features due to waterlogging and are easy to identify and to delimit. But few soil maps exist at an appropriate scale to guide a management and conservation policy for wetlands. As an alternative, maps of the potential waterlogging of soils can be generated using hypotheses about the effect of topography on the soil water regime inspired by Beven and Kirkby's concept of variable source area (Merot *et al.*, 1995). In this work, potentially waterlogged soils were simulated using a threshold value of the topographic index. The procedure was validated by comparing the simulated maps with maps derived from a 1:25,000 soil survey for different catchments. The approach proposed is relevant to model the distribution of intensively waterlogged soils, provided the relationship between the local conditions (essentially bedrock) and the limit value is established. This approach can be improved, taking into account not only the topographic index, but also the relative elevation between the point of interest and the river channel. Crave and Gascuel-Odoux (1996) showed that the spatial distribution of saturated areas near the river is well-explained by a topographic index referring to the downslope conditions. Walter *et al.* (1996) showed that the prediction of the intensity of hydromorphy can be improved by coupling these two upslope and downslope indexes.

Another application of hydrological modelling for characterising the spatial distribution of saturated areas is to study the impact of anthropogenic structures such as hedges on the dynamics of the contributing area. An example is given by Merot and Bruneau (1993): the Breton bocage is a landscape of fields enclosed by a network of hedgerow banks and ditches. One constant feature of this landscape is the presence of banks enclosing bottom lands in the talweg. The topography of a first-order catchment with such a bank was modelled and the effect of the bank on the distribution of the topographic index was simulated. The conclusion is that the saturation zone is confined to the part of the bottom land enclosed by the bank.

MODELLING THE IMPACT OF WETLAND ON EVAPOTRANSPIRATION AND RAINFALL INTERCEPTION

The impact of saturated areas on evapotranspiration at the catchment scale was studied by Quinn and Beven (1991). Traditionally, calculation of the actual evapotranspiration (AE) at the catchment scale is based on potential evapotranspiration (PE):

$$EA = PE \times SM / SM_{max}$$

where:

SM = the current soil moisture

SM_{max} = the maximum soil moisture

Since the topographic index expresses the saturation deficit, it is easy to map the influence of the saturated zone, where saturation deficit is equal to zero, on the spatial distribution of AE.

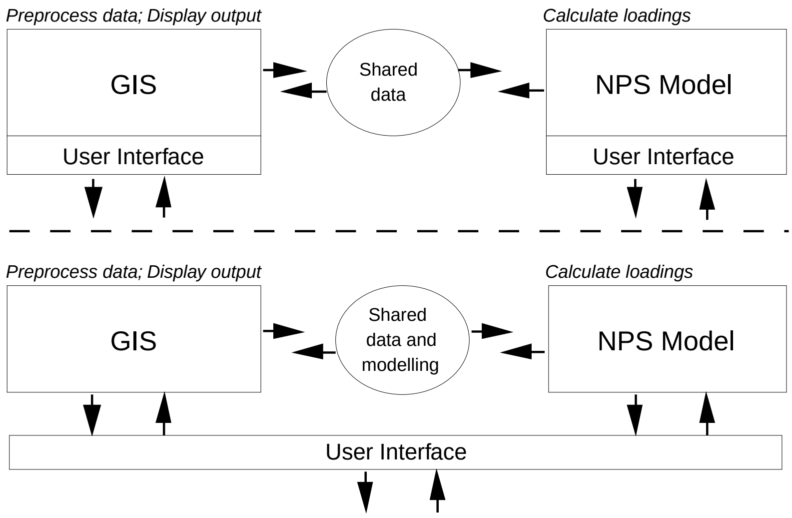
When management of the riparian zone leads to a change in vegetation, for example to increase the leaf area and, consequently, the rainfall interception and transpiration processes, a more complete description of canopy functioning is needed. Yeakley *et al.* (1994), studying the influence of *Rhododendron maximum*, as a keystone riparian species, propose a multi-scaled modelling approach. The model consists of a dynamic canopy interception module and a 2-dimensional hillslope hydrology module (IHDM4: Institute of Hydrology Distributed Model, version 4; Beven *et al.*, 1987). The canopy module takes into account the influence of leaf area index and vegetation type on interception, throughfall and evapotranspiration. The hydrology module is based on the Richards equation. It must be noted that the terrain analysis was not based on a grid network, as in TOPMODEL, but on a contour line-based network (TAPES-C, Moore and Grayson, 1991) that is physically more realistic.

MODELLING THE IMPACT OF WETLANDS ON (BIO)GEOCHEMICAL PROCESSES

The simplest approach is to consider the wetland as a reservoir where mixing processes occur, without any transformation. Linking a mixing model to TOPMODEL, Robson *et al.* (1992) attempted to explain the evolution of chemical composition (via the ANC: Acid Neutralisation Capacity) of flow in a forested catchment, taking into account the saturated area connected to the river. A two end-member mixing model, one end-member for well-buffered deep water and the other one for acidic soil waters allows an adequate simulation of the temporal evolution of river chemical composition, provided that the water of the saturated area has a well-mixed composition. But, as emphasised by the authors and by Durand and Juan Torres (1996), a 2-component model is too limited given the complexity of water paths in a catchment, and specifically in a buffer zone.

More complete models are needed if the complexity of internal geochemical processes inside the wetland has to be taken into account, especially for non-point pollution control. In a recent paper, Poiani and Bedford (1995) consider the interest of GIS-based non-point source pollution modelling for estimating NPS pollution potential or loading rates to wetlands, and to link this information with the spatial characteristics in a catchment (Poiani and Bedford, 1995). They consider two approaches (fig. 3) for coupling NPS pollution models and GIS: 1) linking models with GIS (“linking-model” approach), and 2) modelling within the GIS (“modelling-within” approach). The first approach allows use of sophisticated dynamic hydrological and geochemical models; the second is based on classification, ranking or empirical models, due to the currently poor modelling capabilities of GIS.

Figure 3. Two methods for coupled GIS-simulation model interface:(A) linked-model approach; (B) within-model approach (from Fedra, 1993)



Several critical points must be considered when applying GIS-based NPS pollution modelling to wetlands. Consideration of hydrological inputs to wetlands is the major constraint to be taken into account for modelling: atmospheric input, surface or subsurface runoff, shallow or deep groundwater, streambed or streamflow. For the "modelling-within" approach, new indices such as dilution or denitrification indexes, generally not taken into account in NPS pollution models, are needed. Moreover, this approach has a low temporal resolution and gives qualitative results. Results have to be considered as a first tier screening procedure to target potential problem wetland catchments. The "linking-model" approach seems a more powerful tool. In the distributed models used, the wetland can act as one or several cells, with specific attributes. Nevertheless, problems of model validation occur, due to the difficulty of direct measurements and the complexity of the processes involved, leading to an overparameterisation of the models.

Modelling-within approach

Aurousseau *et al.* (1995) and Aurousseau and Squividant (1995), attempted to estimate the annual nitrogen flux for all catchments in French Brittany (30,000 km^2). 15 to 20% of soils in Brittany are hydromorphic (i.e. the pedological evidence for wetlands in this landscape), generally distributed as narrow strips along channels, or sometimes on badly-drained plateaux. The challenge of this study is to evaluate the impact of wetlands in controlling diffuse nitrogen pollution at a regional scale. Estimation of excess nitrogen inputs was based on a general enquiry at the "commune" level. Water flux was estimated for each catchment using the precipitation and evaporation data provided by the meteorological service. A simple modelling approach within a GIS calculates the annual nitrogen concentration at the outlet of the different catchments. Comparing the estimated concentration to the concentration measured for the same catchments in the regional water quality survey, a reduction of nitrogen losses was found, which was positively correlated to the nitrogen excess. This reduction was mainly attributed to the importance of denitrification (mainly controlled by the level of nitrogen) that occurs in the wetlands as demonstrated at another scale in the same region (Curmi *et al.*, 1996).

An original approach, used by Wolok *et al.* (1989) to understand the chemical control of alkalinity of surface water by saturated areas, can also be linked to this modelling-within approach. The probability of occurrence of saturated areas was computed using the soil-topography index (Beven, 1986) in 145 catchments of north-eastern United States. Indices of flowpath partitioning and soil contact time were derived from the soil-topography index. A good correlation was found between these different indexes and the alkalinity of the lakes draining the catchments. The authors concluded that alkalinity seems to be controlled more tightly by the occurrence of saturated zones than by lithology or precipitation.

Linked-model approach

Poiani and Bedford (1995) reviewed 11 linked-model approaches. The approach for the evaluation of agricultural non point-source pollution proposed by Tim and Jolly (1994) can be considered as a good example of a linked-model approach, although the author classified it as partial integration modelling. After collection, storage and classification in ARC-INFO of numerous information, including land cover and land management data (field number, tracts, field borders), they simulated the impact on sediment yield of different land management strategies using the AGPNS model; these were 1) the establishment of permanent vegetative cover along the streams, 2) contour buffer strips, and 3) a combination of the 2 strategies. By comparing the predicted values to the base condition, recommendations regarding the effectiveness of each land management strategy could be made. However, the simulation was not validated with field data.

Using hydrological modelling to control stormwater in riparian zones

Another property of riparian wetlands is that they can act as floodplains. This leads to the design of wetlands for stormwater pollution control (Wong and Somes, 1995). The main factors to predict are the depth of water, the discharge and the speed of flow. Some models analyse the risk and the vulnerability, depending on land occupation (*inondabilité* method; Barozet *et al.*, 1994) and the hydrologic effectiveness of wetlands (Wong and Somes, 1995). Such models need an accurate topographic map of the valley to show where the flood will inundate. For operational purposes, GIS is used to add information such as anthropogenic or natural obstacles to water pathways and reference information on past floods.

MODELLING THE IMPACT OF HEDGES AS BUFFER ZONES

The potential role of hedges and shelterbelts as buffer zones is often emphasised (Ryszkowski, this volume; Mette and Sattelmacher, 1994). But very few field measurements or modelling attempts have been presented, particularly in temperate climates (Merot and Reyne, 1996), except as a means to reduce wind erosion (De Jong and Kowalchuk, 1995). Hatton and Wu (1995) proposing a scaling theory, to extrapolate individual tree water use to stand water use that attempts to link plant physiology and hydrology, showed a first way of modelling. Basically, modelling capabilities are quite different, considering the hydrological impact of hedges or their geochemical impact. From a hydrological point of view, the two main impacts of hedges are the modification of the water pathways at the soil surface or subsurface and the modification of the water budget by increasing the evapotranspiration. The modification of the surface water pathways can be taken into account by a procedure that transforms a natural drainage network based only on topography to an artificial drainage network that changes the direction of flow paths on pixels crossed by a hedge. Such an approach was used by Zhang (1995) showing the impact of continuous hedges parallel to contour lines on the drainage network. Such hedges disconnect upslope flowlines from the downslope river. Zhang showed, on a catchment where hedge density decreased by a factor of 2 between 1948 and 1994 due to land reclamation, that the fraction of the catchment where flowlines are not directly connected to the river fell from 32% to 14%. The impact of hedges on evapotranspiration could be easily modelled, assuming AE is close to potential evapotranspiration on hedges, provided the DTM used allows specific processes to be modified in pixels crossed by hedges. From a geochemical point of view, a significant effort of characterisation of internal processes inside the hedges and their connections to the surrounding fields remains to be accomplished prior to any modelling effort. Nevertheless a major point to take into account is the location of hedges in the landscape. Hedges parallel to contour lines have probably the higher impact. Hedges whose tree roots are close to the water table seem the most efficient as buffer structures, i.e. hedges surrounding bottom wetlands. Further studies are needed to consider the synergy between wetlands and hedges.

DISCUSSION AND CONCLUSION

Regarding wetlands, two types of model can be distinguished. First, hydrological models specifically based on the variable source area concept provide a good insight into the spatial distribution and the temporal dynamics of riparian wetlands. Unfortunately, few attempts linking such models to biogeochemical models have been undertaken so far. Such models may be well-suited as research tools for the detailed analysis of study cases. Second, the distributed hydrological and Ag-NPSP models that do not currently account for wetland areas have a potential for comparing alternative buffer zone location schemes, providing that an adequate parameterisation of the buffer zone cells is defined. The main limitation of these models relies on the large amount of field data required to constrain and validate the simulations.

At another level, a lot of empirical studies, some of them based on modelling approaches, allow us to take into account the role of wetlands at the catchment scale. The challenge here is to get information in sufficiently contrasting cases to obtain robust relationships and/or to be able to single out the major factors controlling this role.

In all cases, the major constraint is the tremendous variability, in time and space, of the processes that control the functioning of the wetland. Since buffering capacity is related to the fraction of flow flux passing through the active zones of a wetland, a detailed description of the flow paths and of the residence time associated with these flow paths is needed: models will have to take into account the internal variability of the wetlands. It is likely that an adequate modelling should be able to work at a much finer scale in the bottom land than in the rest of the catchment. The variable size of the grid cells in IHDM, smaller downslope than upslope, is an interesting approach in this respect. At large scales, empirical studies and within-GIS model approaches can be used to give a regional range of effectiveness of wetlands as buffer zones and, possibly, of associated gas emissions. More generally, the major restriction on the development and testing of catchment scale buffer zone models is the paucity of comprehensive field studies at this scale. This is even more true regarding other potential buffer structures such as hedges. In this domain, modelling work is only at its very first step and it is based on too scarce field studies.

REFERENCES

Abbott, M., Bathurst, J., Cunge, J., O'Connel, P. and Rasmussen, J. (1986) An introduction to the European hydrological system – système hydrologique européen "SHE" 1: history and philosophy of a physically based, distributed modelling system. *J Hydrol.*, **87**, 45-59.

Ambroise, B., Beven, K.J. and Freer, J. (in press) Towards a generalisation of the TOPMODEL concepts: topographic indices of hydrological similarity. *Water Resources Res.*.

Anderson, M.G. and Burt, T.P. (1980) Interpretation of recession flow. *J. Hydrol.*, **46**, 89-101.

Anderson, M.G. and Kneale, P.E. (1980) Topography and hillslope soil water relationships in a catchment of low relief. *J. Hydrol.*, **47**, 115-128.

Aurousseau, P., Baqué, M.C. and Squividant, H. (1995) *Les basins versants bretons et leur charge polluante.* Rapport et atlas couleur de la Bretagne au 1/900 000. ENSAR, Rennes, 30p. +23 cartes.

Aurousseau, P. and Squividant, H. (1995) Rôle environnemental et identification cartographique des sols hydromorphes de bas fonds. *Revue Ingénieries-EAT, n° spécial 95 Rade de Brest,* C. Cann éditeur, Cemagref éditions. pp 67-74.

Barling, R., Moore, I. and Grayson, R. (1984) A quasi-dynamic wetness index for characterising the spatial distribution of zones of surface saturation and soil water content. *Water resources Res.*, **30**, 1029-144.

Barozet, O., Gilard, O., Givone, P., Sommier, M., (1994) Le modèle de gestion hydraulique du bassin versant de la Bourbre in Crues et inondations. *23 ièmes Journées de l'hydraulique, congrès de la société hydrotechnique de France,* tome, **2**, 727-734.

Beven, K. (1978) The hydrological response of headwater and sideslope areas. *Bull. Hydrol. Sciences,* **23**, 419-438.

Beven, K. (1986) Runoff production and flood frequency in catchments of order n: an alternative approach. In: Gupta, V., Rodriguez-Iturbe I. and Wood E. (Eds.) *Scale problem in hydrology.* D. Reidel, Norwell, Mass. pp 107-132.

Beven, K. (1993) Prophecy, Reality and Uncertainty in distributed hydrological modelling. *Advances in Water Resources,* **16**, 41-51.

Beven, K., Calver, A. and Morris, E. (1987) *Institute of Hydrology Distributed Model,* Institute of Hydrology Report n°98, Wallingford, U.K.

Beven, K. and Kirkby, M.J. (1979) A physically based, variable contributing area model of basin hydrology. *Hydro. Sci. Bull.*, **24**, 43-69.

Beven, K., Lamb, R., Quinn, P., Romanowicz, R. and Freer, J. (1984) TOPMODEL. In: Singh, V.P. (Ed.) *Computer models of watershed hydrology.* Water resources publication. pp 43.

Beven, K. and Wood, E.F. (1983) Catchment geomorphology and the dynamics of runoff contributing areas. *J. Hydrol.*, **65**, 139-158.

Blösch, G. and Sivapalan, M. (1995) Scale Issues in Hydrological Modelling: A Review. *Hydrol. Processes,* **9**, 251-290.

Cappus, P. (1969) Etude des lois de l'écoulement. Application au calcul et à la prévision des débits. Bassin expérimental d'alrance. *La Houille Blanche,* Juillet-Aout 1960 n° A, pp 493-520.

Crave, A. and Gascuel-Odoux, C. (in press) The influence of topography on time and space distribution of soil surface water content. *Hydrol. processes,* **10**.

Curmi, P., Durand, P., Gascuel-Odoux, C., Merot, P. and Walter, C. (in press) Hydromorphic soils and the control of nitrate fluxes: spatial distribution and functional modeling at different scales. Fertilizer research.

De Jong, E. and Kowalchuk, T.E. (1995) The effect of shelterbelts on erosion and soil properties. *Soil Science,* **159**, 337-345.

De Vries, J.J. (1994) Dynamics of the interface between streams and groundwater systems in lowland areas, with reference to stream net evolution. *J. Hydrol.*, **155**, 39-56.

Durand, P. and Juan Torresm J.L. (1996) Solute transfer in agricultural catchments: the interest and limits of mixing models. *J. Hydrol.*, **181**, 1-22.

Ewen, J. (1995) Contaminant transport component of the catchment modelling system SHETRAN. In: Trudgill, S.T. (Ed.) *Solute Modelling in Catchment Systems,* Wiley, Chichester. pp 417-441.

Fedra, K. (1993) GIS and environmental modeling. In: Goodchild, M.F., Parks, B.O. and Steyaert, L.T. (Eds.) *Environmental Modeling with GIS.* Oxford University Press, N.Y., pp 35-50.

Giraud, F. (1992) *Modélisation hydrologique d'une zone humide agricole: perspectives pour l'étude du transport d'azote et de phosphore dans le réseau hydraulique. Cas du marais de Moëze, Charente maritime.* Thèse, Muséum d'histoire naturelle, univ. Rennes 1 (L.E.S.N.M.) 224 p. + annexes.

Hatton, T. and Wu, H. (1995) Scaling theory to extrapolate individual tree water use to stand water use. *Hydrol. processes,* **9**, 527-540.

Hewlett, J.D. and Troendle, C.A. (1975) Non point and diffused water sources: a variable source area problem. In: *Irrigation and drainage symposium,* ASCE, Logan, August, 11-13 75, Watershed management, p46.

Kirkby, M.J. (1978) *Hillslope hydrology.* John Wiley and son, N.Y.

Lesaffre, B. (1988) *Fonctionnement hydrologique et hydraulique du drainage souterrain des sols temporairement engorgés.* Thèse de doctorat., Univ. Paris VI, in Etudes Cemagref, série Hydraulique agricole, n° 4., p 334.

Merot, P. and Bruneau, P. (1993) Sensitivity of Bocage landscapes to surface runoff: application of the Kirkby index. *Hydrol. Processes,* **17**, 167-176.

Merot, P., Ezzahar, B., Walter, C. and Aurousseau, P. (1995) Mapping waterlogging of soils using digital terrain models. *Hydrological Processes,* **9**, 27-34.

Merot, P. and Reyne, S. (1996) Rôle hydrologique et géochimique des Structures Linéaires Boisées: bilan bibliographique et perspectives d'étude. *Et. Tech. Syst. Agraires Dev.,* **29**, 83-100.

Mette, R. and Sattelmacher, B. (1994) Root and nitrogen dynamics in the hedgerow-field interface. Consequences for land use management. In: *proceedings of the 7th Inter. Symp. of CIEC; Agroforestry and land use change in industrialised nations.* Berlin, Humbolt University, pp 275-284.

Mitsch, W.J. (1998) Productivity-hydrology-nutrient models of forested wetlands. In: Mitsch, W.J., Straskraba, M. and Jorgensen, S.E. (Eds.) *Wetland Modelling,* Elsevier, Amsterdam, pp 115-132.

Moore I., Grayson R., 1991 – Terrain-based catchment partitioning and runoff prediction using vector elevation data. *Water Resources Res.,* **27**, 1177-1191.

Moore, I.D., Mackay, S.M., Wallbrink, P.J., Burch, C.J. and O'Louglin, E.M. (1986) Hydrologic characteristics and modelling of a small forested catchment in south-eastern New South Wales. Pre-logging conditions. *J. Hydrol.,* **83**, 307-335.

Moore, I.D., O'Louglin, E. and Burch, G. (1988) A contour based topographic model for hydrological and ecological applications. *Earth Surface Proc. Landforms,* **13**, 305-320.

O'Loughlin, E. (1981) Saturation regions in catchments and their relation to soil and topographic properties. *J. Hydrol.,* **53**, 229-246.

Poiani, K.A. and Bedford, B.L. (1995) GIS-based nonpoint source pollution modeling: considerations for wetlands. *J. of soil and water cons.,* **50**(6), 613-619.

Quinn, P. and Beven, K. (1991) Spatial and temporal predictions of soil moisture dynamics, runoff, variable source areas and evapotranspiration for Plinlimon, Mid-Wales. *Hydrol. processes,* **5**, pp 59-79.

Robson, A.J., Whitehead, P.G. and Neal, C. (1992) Towards identifying sources of subsurface flow: a comparison of components identified by a physically based runoff model and those determined by chemical mixing techniques, *Hydrol. processes,* **6**, 199-214.

Tim, U. and Jolly, R. (1994) Evaluating Agricultural NPS pollution using integrated GIS and hydrologic/water quality Model., *J. Environ. Qual.,* **23**, 25-35.

Walter, C., Antony, V., Thiersault, N., Chaplot, V. and Curmi, P. (1996) Modélisation prédictive de la distribution spatiale des sols hydromorphes de fonds de vallée à partir de critères topographiques et de substrat. In: Walter, C. and Cheverry, C. (Eds.) *Actes des 5 ièmes journées nationales de l'étude des sols.* INRA-ENSA Rennes, pp 93-94.

Ward, R.C. (1982) *The fountains of the deep and the windows of Heaven: perplexity and progress in explaining the response of rivers to precipitation.* University of Hull, pp 30.

Wolok, D.M., Hornberger, G.M., Beven, K. and Campbell, W.G. (1989) The relationship of catchment topography and soil hydraulic characteristics to lake alkalinity in the northern United States. *Water Res. Res.,* **25**, 829-837.

Wong, T.H.F. and Somes, N.L.G. (1995) A stochastic approach to designing wetlands for stormwater pollution control. *Wat. Sci. Tech.,* **32**(1), 145-151.

Yeakley, J.A., Meyer, J.L. and Swank, W.T. (1994) Hillslope nutrient flux during near-stream vegetation removal. I. A multi-scaled modeling design. *Water, Air and Soil Pollution,* **77**, 229-246.

Zhang, Z. (1995) *Hedgerow effect on drainage network.,* Post-doctorat INRA, Science du sol, Rennes, Doc. Int. pp 35.

Part III

The Creation, Restoration and Long-term Sustainability of Buffer Zones

Ecorestoration of riparian forests for non-point source pollution control: Policy and ecological considerations in agroecosystem watersheds

21

B.C. Dickson[1] and D.J. Schaeffer[2]

[1] *Confluence Ecological, P.O. Box 292, Reno, PA 16343, USA*
[2] *Department of Veterinary Medicine/Biosciences, University of Illinois Urbana, IL 61801, USA*

Abstract

Before the turn of the 20th century officials at USDA predicted that water management, through the proper distribution and maintenance of forests, would be essential to the future of farming and strongly advocated the protection and restoration of riparian forests. Subsequent ecological research has significantly expanded our knowledge and understanding of the structure and function of riparian ecosystems, their dynamic nature and linkage to aquatic systems, and the important role they play in landscape ecology. Despite this new information and a growing concern over water pollution problems attributed to agriculture, riparian forests continue to be degraded and destroyed. Studies devoted to understanding specific ecosystem processes (e.g., nutrient dynamics in riparian ecosystems) coupled with ecological research of a more general nature (e.g., land use/water quality relationships) can now be used to develop policies that address specific environmental problems associated with agriculture. These new policies, based in part on restoration ecology, can provide the impetus for a shift away from a concentration on spatially isolated best management practices toward landscape approaches to control nonpoint source pollutants in agricultural regions. The restoration of riparian forests in agroecosystems offers a landscape approach to help control nonpoint source pollution. However, relatively little research aimed at the policy aspects of riparian restoration has emerged, even though ecological information is currently available. Sufficient information now exists to design and restore riparian areas to fully utilise their capabilities in new resource management initiatives to protect water resources. This paper broadly examines the economic, social and ecological aspects of two riparian forest ecorestoration policy options (25 m v 75 m widths) developed to reduce subsurface nitrate movement into surface waters in Illinois (USA) and demonstrates that agroenvironmental policy can and should be based on ecosystem-level management strategies linked to specific ecosystem functional criteria.

INTRODUCTION

The importance of riparian forests for protecting water resources in the United States was recognised by officials at the United States Department of Agriculture (USDA) before the turn of the 20th century. The USDA predicated that water management, through the proper distribution and maintenance of forests, would be essential to the future of farming and strongly advocated the protection and restoration of riparian forests. Subsequent ecological research has supported this position by significantly expanding our knowledge and understanding of the structure and function of riparian forest ecosystems, their dynamic nature and linkage to aquatic systems, and the important role they play in landscape ecology.

Investigations aimed at understanding specific ecosystem processes, coupled with ecological research of a more general nature, can be the foundation upon which policies and management plans are based that target site specific or system-wide processes. This new information, if properly applied, allows planners and resource managers to structure responses to reduce ecosystem dysfunction and promote

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0 © 1997 Quest Environmental.

ecosystem health and maintenance in an efficient, rational manner. Studies of specific ecosystem processes (e.g., nutrient dynamics in riparian ecosystems) coupled with ecological research of a more general nature (e.g., land use/water quality relationships) can now be used to develop policies that address specific environmental problems (e.g., NPS) associated with agriculture.

Sufficient information now exists to design and restore riparian areas to fully utilise their capabilities in new resource management initiatives to protect water resources. This paper broadly examines the economic, social, and ecological aspects of two riparian forest ecorestoration policy options (25 m v 75 m widths) developed to reduce subsurface nitrate movement into surface waters in Illinois (USA), and demonstrates that agroenvironmental policy can and should be based on ecosystem-level management strategies linked to specific ecosystem functional criteria.

RIPARIAN FORESTS AND NPS CONTROL

The successful application of riparian systems for NPS control and water quality improvement requires an understanding of the structural and functional ecology of these unique ecosystems, the ecological services they provide, and how best to utilise them in agroecosystem watersheds. Many studies have shown that relatively narrow forest buffers, often 30 meters wide or less, can trap sediment and filter or transform chemicals emanating from upland terrestrial environments before they reach surface waters (Peterjohn and Correll 1984; Lowrance *et al.*, 1984, 1985; Jacobs and Gilliam, 1985; Pinay and Décamps, 1988; Cooper, 1990; Kovacic *et al.*, 1990; Taylor e*t al.*, 1990; Ambus and Lowrance, 1991). Research continues to support and refine these findings (see papers in this volume) and is providing new information for riparian forest ecorestoration design. However, the costs and benefits associated with the development and implementation of watershed-based riparian forest restoration projects, the changes in landuse and expected water quality improvement, and the acceptance or rejection by farmers are unclear. In short, many issues regarding the feasibility of riparian forest restoration in agroecosystems remain.

ILLINOIS CASE STUDY

Thirteen watersheds in Illinois (USA) were examined in an attempt to determine if riparian forest restoration was feasible. All of the watersheds were located in the Central Corn Belt Plains Ecoregion (Omernik and Gallant, 1988), had predominantly agricultural land use (row crops) and showed relatively high concentrations of nitrate (mean = 7.2 mg l^{-1}; se = 0.113, n = 1648) that had not changed significantly between 1977 and 1992 despite conservation programs. In addition, regression models showed that, for all watersheds, (aggregated) annual total fertiliser nitrogen application accounted for most of the variability in seasonal (December through June) stream nitrate concentration (R^2 = 0.87) (Dickson, 1995).

Watershed morphology and riparian forest assessment

Drainage density for all watersheds in this study averaged 1.06 ± 0.39 mi/mi^2 (i.e., linear stream miles per square mile of area) which is very close to the "most typical" value of 1 mi/mi^2 given by Omernik and Gallant (1988) for the Central Corn Belt Plains Ecoregion. As a percentage of total stream network length, areas having riparian forest coverage was relatively low using two riparian forest assessment methods. Analysis of Illinois Stream Information System (ISIS) data consistently showed greater riparian coverage than that developed from geographic information system (GIS) measurements. Riparian forest extending ~ 25 meters from the stream bank averaged 16.3 ± 8.74% (range 5.1% to 32.9%) in the watersheds according to ISIS; GIS generated data from 1:100,000 USGS maps showed a much lower average percentage coverage at 5.2 ± 3.15 (range 1.2% – 13.0%).

The GIS assessment also provided data that was used to characterise basic watershed morphology, the distribution of riparian forests in the drainage networks and the nature of the riparian forests in the watersheds (Table 1). The percent of coverage, length and distribution of riparian forest patches found throughout the stream networks was startling. Moreover, GIS data showed that riparian forest "patches" average only 451 m in length (n = 700) in the thirteen watersheds. Average riparian forest length increases with stream order but is still quite limited: first order streams averaged 354 m, second order streams 402 m, third order 531 m, fourth order 579 m and fifth order segments 660 m. The percentage of riparian forest coverage in the watersheds also increases by stream order. A scant 2.4% of first order streams have riparian forest coverage, yet first order streams represent over 63% of all stream miles in the watersheds. Riparian forest coverage in second order streams averaged 5.8%, third order 13.6%, fourth order 15.8% and fifth order streams roughly 21%.

Table 1. Mean length, percent of total network length, riparian forest length, and percent of riparian coverage by stream order for all watersheds.

Stream Order	*Mean Total Length (km)*	*Percent of Network*	*Riparian Forest Length (m)*	*Percent of Length with Coverage*
1	2.74	63.48	354	2.40
2	3.62	19.76	402	5.79
3	9.54	12.24	531	13.60
4	11.09	5.50	579	15.79
5	22.40	4.50	660	21.24

Riparian forest restoration potential

The percentage of each watershed eligible to be restored to riparian forest for both policy options (25 m v 75 m) varied among the watersheds and reflected the same basic difference as previously observed between ISIS and GIS estimates of riparian coverage. Considering the differences in riparian assessment methods, the percentages of riparian area eligible for forest restoration under the 25 m policy option are quite small, the average ranging from 1.2 to 3.1% of the total watershed area. The average percentage for the 75 m policy ranged from 4.5% to 9.3%.

The quantity of restorable land/farm in each of the watersheds was considered to be especially important because the decision to participate in this type of conservation program is ultimately made on the farm. On the average farm (mean = 145 ha) under the 25 m forest restoration policy, the average amount of eligible land ranged from 1.74 to 4.49 ha/farm; for the 75 m option the average area expanded to 6.52 to 13.52 ha/farm. These figures reflect the differences in the quantity of riparian forest already found in the watersheds and the slight disparity in drainage density.

Projected economic benefits and costs: regional watershed assessment

Calculations of regional benefits and costs associated with both forest restoration options (aggregated for all watersheds) varied widely. Estimated economic benefits (cost-share payments to farmers, land rental payments to farmers, timber, value of water quality improvement, value of wildlife habitat) exceeded economic costs (implementation costs, lost production) under both policy scenarios when set to net present value (assumes implementation on all eligible acres with contracts lasting 15 years in a Conservation Reserve type program). Potential benefits under the 25 m policy exceed costs by $53 to $162 million depending on whether the estimates were developed from ISIS or GIS data and land rental rates (LRR) of $90, $120 and $150 per acre; the best estimate (geometric mean) of benefits ranged from $85 to $100 million. Benefits exceed costs for the 75 m restoration policy option as well; approximately $189 to $466 million in benefits could be expected with a best estimated range of $273 to $323 million (Dickson, 1995).

Projected economic benefits and costs: farm level assessment

Implementation of either the 25 m or 75 m riparian restoration policy would generate substantial regional benefits as discussed previously. A significant portion of these benefits (cost-share payments and annual land rental payments) would be distributed among farmers participating in the 15-year conservation program. Economic benefits received by individual farmers participating in the riparian forest restoration program would depend on the quantity of eligible land on their farm. The estimated costs and benefits of program participation for the average farm under both policy options show that participation is justified. Net benefits of program participation associated with the 25 m restoration policy ranged from $1,099 to $2,836/farm at a land rental rate of $90/acre, $2,533 to $6,537/farm at a land rental rate of $120/acre and $3,967 to $10,239/farm at a land rental rate of $150/acre. Benefits of program participation associated with the 75 m restoration policy ranged from $1,313 to $2,726/farm at a land rental rate of $90/acre, $6,683 to $13,866/farm at a land rental rate of $120/acre and $12,053 to $25,006/farm at a land rental rate of $150/acre. These figures are representative of the benefits expected considering the assumptions of this analysis. However, benefits from water quality improvement, wildlife habitat creation, and timber could be gained by participating farmers (although not included in the farm level assessment) making these estimates conservative.

CHARACTERISTICS AND IMPLICATIONS OF FOREST RESTORATION POLICY OPTIONS

The results of both the water quality assessments and trend analysis support the development of riparian forest restoration policies to improve water quality in the Central Corn Belt Plains Ecoregion. Furthermore, the results of the riparian forest assessments conducted in the thirteen study watersheds clearly show that the majority of stream miles could be restored. Consequently, such a policy would be well suited to this ecoregion.

The general absence of riparian forests and the short length of streamside forest patches found in these watersheds are indicative of a heavily developed, fragmented riparian ecosystem. These findings also suggest that on first, second and third order streams the removal of riparian forests is a standard practice where agriculture is the dominant land use. Conversely, with increasing stream order riparian forests become a more viable land use in these Central Corn Belt Plains Ecoregion watersheds. For example, the percentage of riparian forest on fifth order streams is ten times greater than first order streams. This is probably a consequence of a broadening flood plain that discourages crop production. At some point levees and or drainage become practical and riparian forests are again converted to agricultural production.

Even though benefits are anticipated at the regional watershed level the potential benefits calculated at the farm level are probably more meaningful to farmers than regional estimates, for it is here that the decision to participate in conservation programs is ultimately made. If benefits cannot be expected for the average farm in these watersheds, then little chance of enrolment exists. Estimates of the quantity of eligible land/farm ranged from 1.74 ha to 4.49 ha (best estimate 2.79 ha) for the 25 m policy option and 6.51 to 13.52 ha (best estimate 9.39 ha) for the 75 m option. Based on $90, $120 and $150 per acre land rental rates, and without including timber value and ecological benefits, modest per farm net benefits are predicted. The most conservative estimate for both the 25 m and 75 m policy option (ISIS data and $90/acre LRR) show benefits of $1,099 and $1,313 per farm respectively.

At the farm level the 25 m policy option provides a better return per acre than the 75 m option. Net benefits per acre at the $90, $120 and $150/acre rental rates for the 25 m policy option are $255, $589, and $922 per acre respectively whereas the net benefits under the 75 m option are $82, $415 and $748/acre. This difference is attributable to improved soil productivity further from stream banks and its greater profitability. As a result, opportunity cost for riparian farm land enrolled under the 25 m policy is lower than that of the 75 m policy.

The relationship between land rental rate and opportunity cost is critical because farmers, faced with the uncertainty of future crop prices, must believe that they can recover the opportunity cost of enrolling their land from the LRR. Production losses from enrolled cropland directly affect the benefits that could be expected by farmers. Based on average net income/acre for crops between 1983 to 1992, a LRR of $67.10/acre is necessary to break even for the 25 m policy option; a LRR of $82.62/acre is the break even point for the 75 m option. If 1992 net income/acre is used instead of the 1983 to 1992 average, then LRRs must increase because of the greater influence of lost production in the calculation of net benefits/acre. The LRR for the 25 m policy option must increase to $106.21/acre to produce any net benefits to farmers. Similarly, the LRR for the 75 m policy would need to increase to $117.10/acre.

Even though the farm level economic analysis showed benefits to farmers from enrolling in the program under either the 25 m or 75 m option some may wish to enrol only part of their eligible ground while others may not enrol any eligible land in the program. One deterrent to enrolment may be the perception by farmers that they are sacrificing too much of their available producing land, particularly on small farms. This hypothesis, whether well-founded or not, gives a decided advantage to the 25 m policy. On the average farm in these watersheds, between 1.74 and 4.49 ha are eligible under the 25 m policy whereas 6.52 to 13.52 are eligible under the 75 m policy. This equates to 1.2 to 3.1% of total area on the average farm (145 ha) for the 25 m policy compared to 4.5 to 9.3% for the 75 m policy. The percentage associated with the 25 m policy would probably not inhibit enrolment because average annual crop reduction under the U.S. Feed Grains Program Acreage Reduction Program (ARP) has been well above this range since 1982 (Mercier, 1990). The average ARP for corn between 1982 and 1994 was 10.8% (± 5.6). Land removed from production under the 75 m policy option (4.5 – 9.1%) may appear to be excessive to farmers in relation to previous and anticipated corn ARP percentages, although this range is still below the thirteen year average (Dickson, 1995).

A comparison of the value of ecological benefits attributable to implementation of the 25 m and 75 m policy options at the regional watershed level show that the benefits associated with water quality improvement are significantly greater than those generated by wildlife habitat (Dickson, 1995). Using the most conservative estimate of ecological benefits (ISIS data), more than $36 million would be realised from implementation of the 25 m policy and roughly $135 million for the 75 m policy. The total economic benefits from re-establishing riparian forests would be much greater than those estimated herein. Only two ecological benefits, water quality and wildlife habitat, are included in this analysis.

PREDICTED STREAM NITRATE REDUCTION

The effectiveness of riparian forest buffers to reduce nitrate has been demonstrated in several studies including one in the Central Corn Belt Plains Ecoregion (Kovacic *et. al.*, 1990). Forest buffers ≥ 25 m are expected to reduce by approximately 85% (average based on several studies) the nitrate passing through them in the study watersheds. The amount of drainage passing through riparian forests is assumed to be the same as the percentage of area in riparian forest cover with two exceptions. In areas where subsurface drainage exists (35% of Illinois farmland), no reduction in nitrate discharge is expected to occur even if these areas were reforested and existing riparian forests (~ 10%) would not further reduce nitrate discharge. This leaves 55% of eligible acres capable of reducing nitrate discharge to streams. Based on these assumptions, if a large percentage of eligible riparian land was restored to riparian forest in the study watersheds, significant reductions in nitrate concentration are expected (Table 2).

Table 2. Predicted nitrate concentration by percent coverage following reforestation of riparian areas at least 25 m wide in Central Corn Belt Plains Ecoregion watersheds. (Predicted concentration calculated from initial concentration of 7.2 mg l^{-1}.)

Percent Coverage	*Reduction Efficiency*	*Percent Reduction*	*Predicted Nitrate Concentration*
0.20	0.85	0.170	6.0
0.30	0.85	0.255	5.4
0.40	0.85	0.340	4.8
0.50	0.85	0.425	4.1
0.55	0.85	0.468	3.8

POLICY CHOICES AND LANDSCAPE DEVELOPMENT

Given the choice between either of the two policy options (if offered in a conservation program) it is more likely that a larger percentage of eligible acres would be enrolled by farmers under the 25 m option. The resulting spatial arrangement of a preference for the 25 m versus the 75 m option may actually contribute to greater connectivity of riparian patches throughout the watersheds. A greater number of smaller patches with a high degree of connectivity may produce more benefits than a smaller number of disconnected, isolated self-sustaining patches with greater biodiversity. Ultimately, enrolment would be directed by a mix of ecological and economic factors and how they are perceived by farmers. A situation with relatively narrow reforested riparian strips (i.e., 25 m) in the upper reaches of stream networks that gradually increase in width (eventually exceeding 75 m) in conjunction with wider flood plains downstream is most probable.

CONCLUSION

Riparian restoration policies have the potential to reduce nutrient losses to surface waters while removing a relatively small percentage of high-risk land from crop production. Targeting riparian areas for forest restoration along stream networks in these Illinois (USA) watersheds has the potential to lower average nitrate by nearly half if at least 55% of all eligible riparian lands were enrolled under the 25 m policy option. Both policy options produce substantial regional water quality and habitat benefits but on a per acre basis the 25 m option is superior. Over a range of land rental rates ($90, $120, $150/acre) modest benefits are realised on the average farm in these watersheds without including potential timber value and ecological benefits consumed by landowners.

The low percentage of riparian forest coverage and small size of forest patches indicates a severely fragmented, dysfunctional riparian ecosystem having limited interaction with adjacent aquatic systems. If a riparian restoration program were initiated, it is reasonable to expect that the health of these agroecosystem watersheds would improve benefiting both landowners and society. However, it must be remembered that ecorestoration policies must be integrated with better on-farm management and more sustainable agricultural practices in order to minimise environmental impacts.

This paper also demonstrates that policies of this type can be developed from sound ecological information and designed to match conditions at a local or ecoregional level, that riparian forest restoration is a viable policy instrument for NPS mitigation due to a unique mix of ecological and economic factors, and agroenvironmental policy can be based on ecosystem-level management strategies and linked to specific ecosystem functional criteria.

REFERENCES

Ambus, P. and Lowrance, R. (1991) Comparison of denitrification in two riparian soils. *Soil Sci. Soc. Am. J.*, **55**, 994-997.

Cooper, A.B. (1990) Nitrate depletion in the riparian zone and stream channel of a small headwater catchment. *Hydrobiologia*, **202**, 13-26.

Dickson, B. C. (1995) *Ecorestoration of Riparian Forests for Nonpoint Source Pollution Control: Policy and Ecological Considerations in Illinois Agroecosystem Watersheds.* PhD Thesis, University of Illinois, Urbana, IL.

Jacobs, T.C. and Gilliam, J.W. (1985) Riparian losses of nitrate from agricultural drainage waters. *J. Environ. Qual.*, **14**, 472-478.

Kovacic, D.A., Osborne, L.L. and Dickson, B.C. (1990) *The Influence of Riparian Vegetation on Nutrient Losses in a Midwest Stream Watershed.* Water Resources Center, Urbana, IL.

Lowrance, R.R., Todd, R.L. and Asmussen, L.E. (1984) Nutrient cycling in an agricultural watershed.: Phreatic movement. *J. Environ. Qual.*, **3**, 22-27.

Lowrance, R.R., Leonard, R.A. and Asmussen, L.E. (1985) Nutrient budgets for agricultural watersheds in the southeastern coastal plain. *Ecology*, **66**, 287-296.

Mercier, S. (1990) Corn. In: Linwood Hoffman (Ed.) U.S. *Feed Grains: Background for 1990 Farm Legislation.* Agriculture Information Bulletin 604. Commodity Economics Division, Economic Research Service, U.S. Department of Agriculture, Washington, D.C.

Omernik, J.M., and Gallant, A. (1988) *Ecoregions of the Upper Midwest States.* EPA/600/3-88-037. U.S. Environmental Protection Agency, Corvallis, Oregon.

Peterjohn, W.T. and Correll, D.L. (1984) Nutrient dynamics in an agricultural watershed: Observations on the role of a riparian forest. *Ecology,* **65**, 1466-1475.

Pinay, G., and Décamps, H. (1988) The role of riparian woods in regulating nitrogen fluxes between the alluvial aquifer and surface water: A conceptional model. *Regul. Rivers Res. Manage.,* **2**, 506-516.

Taylor, J.R., Cardamone, M.A. and Mitsch, W.J. (1990) Bottomland hardwood forests: Their functions and values. In: Gooselink, J., Lee, L.C. and Muir, T.A. (Eds.) *Ecosystem Processes and Cumulative Impacts: Illustrated by Bottomland Hardwood Wetland Ecosystems.* Lewis Publishers, Chelsea, MI.

22

Economically viable buffer zones – the case for short rotation forest plantations

D. Riddell-Black[1], G. Alker[1,3], C.P. Mainstone[2], S.R. Smith[1] and D. Butler[3]

[1] *Soil, Waste and Groundwater Group, WRc plc, Medmenham, Henley Road, Marlow, UK*
[2] *Ecological Assessment Group, WRc plc*
[3] *Department of Civil Engineering, Imperial College London, College Road, London, UK*

Abstract

Short rotation forestry crops (SRF), more frequently known as short rotation coppice in the UK, are an expanding feature in the European agricultural landscape. The crop has primarily been developed as a renewable fuel source but also has uses as a chip and fibreboard raw material. Though hectarages are still currently small, support systems are in place to promote its development and between 20 000 and 30 000 hectares are projected to be planted in the UK alone in the next 5 to 8 years.

The high productivity, 10 to 20 oven dry tonnes per hectare and year, and frequent removal of nutrients in harvested stems, differentiate this crop from more traditional wood production systems. Regular fertiliser inputs are a feature of its management. Consequently, there is great interest in the use of the crop to manage liquid wastes such as dairy shed washings, silage liquors, landfill leachate, sewage effluent and industrial processing effluents, such as from meat works. Work in Sweden has focused on its use for sewage and groundwater renovation, while research in the UK has been considering its use in the recycling of sewage sludge.

The dense root network, high nutrient uptake and low management inputs of SRF suggested that it may be effective at the removal of nutrients and the assimilation of organic pollution if located as a buffer zone. Two approaches may be adopted, point source and diffuse source interception. Where there is a risk of occasional losses from storage or handling facilities a plantation could be established as a barrier to prevent pollution transport to vulnerable water courses. Alternatively, a plantation may be located at margins to watercourses to intercept diffuse nutrient and pesticide run-off and spray drift. In such a location it would have the additional benefits of stabilising the riverbank against erosion and restricting the direct access of livestock to the stream bed (which may otherwise result in significant direct inputs of animal excreta). The principal advantage of SRF as buffer zone for the farmer is that an income can still be obtained from land which, if dedicated to water protection, would otherwise not contribute to the farm finances.

WRc is currently investigating the nutrient uptake capacity of a range of coppice species and clones of willow and poplar. Two techniques, lysimeter studies and a hydroponic growing system, are being used to separate the role of the tree from that of the soil in the renovation of nutrient rich liquors. It is hoped that the study will identify tree varieties which have a high renovative capacity which can then be tested in field locations. This paper describes the study method and results to date and considers how SRF production can be managed to operate as buffer zones.

INTRODUCTION

The use of buffer zones to protect water bodies from the effects of man's activities within a catchment is a system of land management with the specific objective of environmental protection. There are potentially a wide range of methods of applying buffer zones and the definition of what constitutes a buffer zone appears to be fairly liberally interpreted. However, in general terms it can be described as an area of land, often a boundary, located between a source of pollution and a temporary or

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0

permanent sink for that pollution, which is designed to reduce or neutralise the effects of the pollution. The land take required for a buffer zone to be effective will vary with location, the nature of the pollutant and of the receiving water.

A wide range of physical, biochemical and biological processes contribute to the efficacy of a buffer zone and the pathways of substances which enter them are complex and not well understood. One concern that is as yet unanswered is whether there will come a point when a buffer can no longer absorb the pollutants entering it and may ultimately become a source, rather than a sink for contaminants. This has been demonstrated in connection with phosphorus (P) which, once absorbed by plants within a buffer, potentially becomes more available when that plant dies and degrades than it was when it entered the buffer zone. Hence, a buffer zone can become a net exporter of available P (Dillaha *et al.*, 1989).

The effective life of a buffer zone may be prolonged if the rate at which it moves towards saturation can be slowed through the frequent removal of contaminants. In the context of an agricultural catchment, this largely will apply to N and P inputs, pesticides usually being degraded by soil microbial processes within the buffer zone. Where conditions permit, denitrification of NO_3-N to N_2O and N_2 gas will prevent N accumulation. However, in a wider range of conditions, the harvesting of plant material from a buffer zone can contribute substantially to the removal process and is probably the only cost-effective way of achieving nutrient removal to sustain buffer zone efficacy. Where the harvested material has an economic value, the likelihood of landowners managing the task effectively is greatly improved.

Short rotation forestry is a low input wood production system which involves the frequent removal of plant biomass from the site of production and with it the nutrients it contains. High density planting, the semi-permanent root system and relatively high nutrient take up characteristics all contribute to the suitability of short rotation forestry as a buffer zone crop. The principal advantage of short rotation forestry as a buffer zone crop for the farmer is that an income can still be obtained from land which, if dedicated to water protection, would otherwise not contribute to the farm finances.

A study is currently being undertaken to investigate the nutrient uptake capacity of a range of short rotation forestry species. The techniques being employed are designed to separate the role of the plant from that of the soil in the renovation of nutrient rich liquors in order to ascertain the relative contribution of each component of the system. This paper provides some background to the development of short rotation forestry, describes the study method and presents some initial results.

RATIONALE

Short rotation forestry (SRF) crops are an expanding feature in the European agricultural landscape as a consequence of pressure to reduce food surpluses. The crop is being developed primarily as a renewable fuel source but also may be used as a chip and fibre-board raw material. Short rotation forestry has been identified as the most promising renewable energy source in the UK in terms of the potential size of its contribution to meeting energy demand and its closeness to commercialisation (HMSO, 1992). In Sweden, 18 000 ha of willow SRF have been planted and about 2000 ha continue to be planted annually. Short rotation plantations of poplar have been identified as offering the greatest potential for renewable energy development in the Netherlands (NOVEM, 1992). It is probable that a significant area of SRF will be planted in Europe in the short to medium term.

Plantations consist of densely planted trees, 10 000 to 18 000 per hectare, which are repeatedly cut over rotations of 2 to 5 years, depending on species and target market. It is anticipated that individual plantations will have a productive life of 25 to 30 years. Species most commonly in use for fuel in the UK are willow and poplar.

There are a number of reasons why SRF may be appropriate for incorporation into water protection buffer zones. The perennial nature of the crop means that the land it occupies is effectively taken out of production with regard to cultivation for long periods. Management inputs to SRF are low when compared with agricultural crops, consisting primarily of weed control by chemical or mechanical means in the two year establishment period. Thereafter, in a well-established plantation, no further herbicide application will be required for 25 to 30 years. Harvesting activities are greatly reduced compared with normal agricultural systems. Trees continue to take up nutrients late in the season when the potential for nitrate leaching losses is highest. As plantations mature, a woodland herbaceous understorey develops which will contribute to surface roughness and hence promote infiltration of surface flow. Up to 6 dry tonnes per hectare of leaf litter can be contributed to soil organic matter annually (Ericsson *et al.,* 1992), improving soil structure and hence infiltration capacity. The high proportion of crop margin in SRF may contribute to integrated pest management by providing hedgerow-type habitat for foraging birds and predatory insects which may facilitate a reduction in the use of pesticides. However, the corollary of this may be that a weed bank is created which seeds into the productive areas that the SRF borders. Hence, the balance of benefits against costs needs to be assessed according to the needs of a specific location.

Plantations can be designed to optimise their efficacy as buffer zones. Contour planted rows will maximise the interception of surface water flow as slight ridging develops along the row of root stools. Grassed access alleyways will further contribute to surface roughness, promoting infiltration and trapping particulates. Short rotation harvesting means that the crop is maintained in a state of maximum nutrient accumulation potential. A typical harvest cycle of three years will result in only one third of the crop being harvested annually, minimising the habitat implications of the harvesting operations. Hence there is never an occasion when there is no standing crop in the buffer.

Short rotation forestry is increasingly being incorporated into liquid waste management systems, either as a final polishing stage, or as an alternative to chemical and biological treatment. In Sweden, 12 ha of willow have been planted to remove N and P from sewage effluent following secondary treatment (Hasselgren, 1994). Single stem poplar are being used for treating landfill leachate in Oregon, USA (Licht, 1994). In New Zealand, the use of short rotation *Eucalyptus* for treating a range of aqueous wastes is increasing. Effluents treated in this manner include liquors from meat processing works, a wool scour plant and dairy shed washings (Sims *et al.,* 1994, Riddell-Black *et al.,* 1996).

The dense root network, high nutrient uptake and low management inputs of SRF suggest that it may be effective scavenger for mobile nutrients if located as a buffer zone. Two approaches may be adopted, point source and diffuse source interception. Where there is a risk of occasional losses from storage or handling facilities a plantation could be established as a barrier to prevent pollution transport to vulnerable water courses. Alternatively, a plantation may be located at margins to watercourses to intercept diffuse nutrient and pesticide run-off, spray drift and particulate movement in overland flow. In such a location it would have the additional benefits of stabilising the riverbank against erosion and restricting the direct access of livestock to the stream bed, which may otherwise result in significant direct inputs of animal excreta.

This study is investigating the ability of SRF to treat nutrient rich effluents. The capacity of soil to remove nutrients and solids from effluent through physical and biochemical processes is already well known. However, the relative contribution of plants to the renovation is not well-understood. It is necessary to ascertain the level of contribution that the trees make in order to establish the extent to which tree management will influence buffer zone efficacy and can be manipulated to enhance that efficacy.

There are some 300 species of willow and a considerably greater number of clones. This provides a huge resource in terms of selecting varieties which are suitable for use in high nutrient situations such as effluent renovation schemes or buffer zones. Previous work has shown substantial differences between clones in both their productivity and nutrient uptake characteristics (Ericsson *et al.,* 1992, Riddell-Black, 1996). An initial screening of a large number of clones is necessary to identify those with greatest nutrient removal potential, either in terms of biomass production or stem nutrient concentration, or both. Ideally, in habitat terms a woodland buffer zone of native species would be preferable to bred clonal plants. Mixed clonal plantations are the norm for disease prevention purposes. However, differential growth makes it difficult to mix species without sacrificing productivity and ultimately the competitiveness of faster growing species would lead to an overall thinning of the plantation and hence potentially reduced efficacy as a buffer zone.

The study has four experimental components:

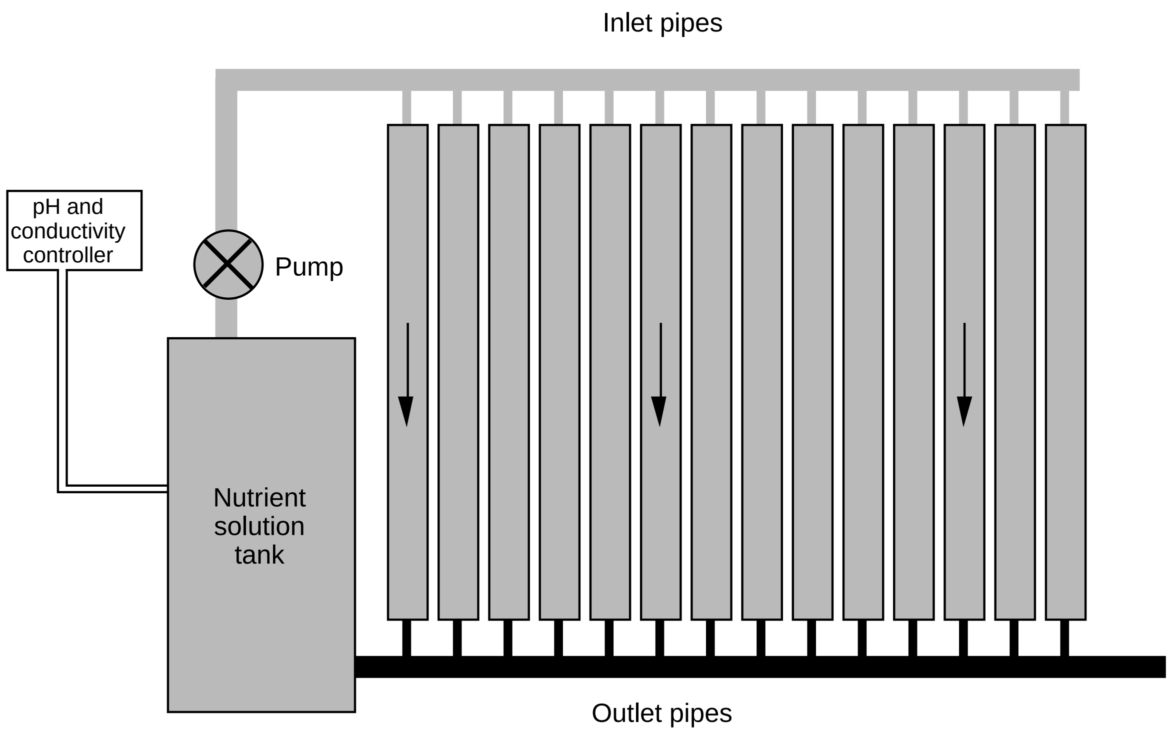
- rapid screening of clones for nutrient uptake and biomass production using a soil-less system
- testing of screened clones survival in and nutrient uptake from waste effluents
- lysimeter study of willow irrigation with landfill leachate and sewage effluents
- root decay and soil incubation tests of N release characteristics of plant leaf and root biomass

The initial screening stage is underway; these trials will allow the determination of clones suitable for use in effluent phytoremediation and buffer zones. Specifically, a suitable clone will be one that can accumulate a large quantity of N and P within its biomass, the allocation of which is such that the greater proportion will be in the harvestable portion, that is the stems. This paper describes the method operated and the results obtained from the initial trial run.

METHOD

The rapid screening apparatus consists of 14 plastic channels which carry circulating nutrient solution pumped from a central tank (Figure 1). The whole unit is housed in a controlled environment glasshouse. The nutrient solution is made to standard formula (Hewitt, 1966) and maintained by a pH and conductivity controller to ensure constant conditions. The solution also contains an ammoniacal nitrogen source, which is the principal N form in landfill leachate and which is circulated in a single system to all the channels at the same flow rate to ensure that each of the clones receive the same quantity of nutrients.

Figure 1. Diagrammatic representation of the screening apparatus



Willow cuttings 120 mm in length are established in rockwool blocks in combination with perlite in de-ionised water, then selected for uniformity and placed in the channels with the root system partially submersed in the nutrient solution. Five cuttings are grown in each channel and each clone occupies 2 channels, such that there are 10 replicates of each clone. A reference clone is grown in every run to check consistency between trials and enable any variation in conditions between test runs to be detected, hence 6 clones are tested in each run.

The test has a duration of 5 weeks during which a number of growth parameters are measured, namely leaf number and area, stem diameter at a fixed height and shoot height. At the end of each run, root, shoot and leaf biomass are destructively determined and N and P content of each component are analysed to determine uptake and partitioning between plant parts. The initial screening of clones is at an early stage; the plant tissue concentrations of N and P from the first test run are presented in Table 1.

Table 1. N and P concentrations of plant components after 5 weeks' growth in the screening apparatus (mg g^{-1} dry matter)

	N				*P*			
Willow clone	*Stem*	*Leaf*	*Root*	*Cutting*	*Stem*	*Leaf*	*Root*	*Cutting*
Bowles Hybrid	16.4	36.7	24.2	6.3	3.5	6.9	17.4	1.0
Calodendron	11.8	36.0	17.1	4.0	3.2	5.2	18.3	1.0
Dasyclados	13.2	35.9	23.3	5.5	3.2	5.4	17.4	1.1
Gigantea	16.9	39.4	27.0	5.7	3.7	9.7	18.4	1.1
Mullatin	14.4	40.8	23.8	4.4	3.0	6.5	18.1	0.7
Q83	12.4	39.5	24.7	4.5	3.5	5.1	17.3	1.5
Vim 683	25.2	43.0	28.6	6.1	3.9	11.6	14.4	1.2

DISCUSSION

Typical stem concentrations of N and P in trees grown in the field are 3 to 4 mg g^{-1} dry matter and 0.8 to 1.1 mg g^{-1} dm respectively (Ericsson *et al.*, 1992). Hence, the annual nutrient export in harvested stems by a crop yielding 10 oven dry tonnes per hectare and year (odt ha^{-1} yr^{-1}) will be 30 to 40 kg N ha^{1} and 8 to 11 kg P ha^{-1}. Nitrogen concentrations measured in two-year-old stems of willow coppice grown at nutrient rich sites have ranged from 6.5 to 10.7 mg g^{-1} depending on willow variety (Riddell-Black, 1996). Plant tissue concentrations measured from glasshouse tests are frequently higher than those found in the field. Additionally, the stems grown within the screening system are immature and hence typically will contain higher N concentrations than mature stems. Immature stems also contain a higher proportion of nutrient rich bark than do mature stems. These phenomenon can be seen in the difference in the N and P concentrations between the stem and the cutting, the cutting being the mature stem material used to establish the crop (Table 1).

The results obtained from the screening test must be referenced against stem concentrations of the same clones measured at a range of sites to establish the extent to which the glasshouse test accurately reflects the performance of the clones in the field. This will also indicate whether the ranking of clones relative to each other is consistent between field and glasshouse conditions. If it is assumed, for the purposes of illustration, that the stem concentrations of N and P given in Table 1 are indicative of field conditions with high nutrient input, such as an effluent treatment system or buffer zone with high runoff concentrations, the clone *Salix viminalis* '683' would remove 252 kg N ha^{-1} yr^{-1} given a typical annual stem biomass production of 10 odt ha^{-1}. In contrast, *Salix triandra x viminalis* 'Q83' would export only 124 kg ha^{-1} yr^{-1} at the same productivity level. Equivalent P removal would be 39 and 35 kg P ha^{-1} yr^{-1} respectively.

At the outset it was stated that the purpose of harvesting plant biomass from buffer zones was to limit the accumulation of nutrients which ultimately may reduce or negate the efficacy of the buffer zone. Typical losses of N and P from different rural land uses are given in Tables 2 and 3. A typical catchment will comprise many different land uses and the total amount of N and P delivered to a water course will be determined both by the land uses and the size of the catchment. In crude terms, if a catchment of 10 km^2 (1000 ha) were comprised entirely of grassland delivering 15 kg ha^{-1} N and 2 kg ha^{-1} P, the draining water course would be in receipt of 15 000 kg N and 2000 kg P annually. If a SRF buffer zone were to entirely remove this quantity, a 100 ha buffer zone would be required, assuming a stem concentration of 15 mg g^{-1} N and yield of 10 odt ha^{-1} yr^{-1}. This implies that the buffer zone would require to be in a ratio of 1 to 10 of the land area it was to treat, if it were to export the amount of N and P equivalent to that delivered to the buffer. This is clearly not practical or economic and highlights the difficulty of dedicating sufficient land to buffer zones for them to be effective. If there were to be 1 ha of SRF for every 100 ha of catchment, stem N concentrations would need to be 150 mg g^{-1}, which is highly improbable, if annual yield were 10 odt ha^{-1}. Despite the high biomass production and hence nutrient removal potential of SRF, nutrient export by this means alone will not be sufficient to alleviate the effects of polluted drainage. Hence, buffer zones must be designed to allow a range of processes to operate.

Table 2. Nitrogen export coefficients for rural land uses (after Johnes *et al.*, 1994)

Land Use	*Location*	*Annual export rate (kg N ha^{-1})*	*Reference*
Unfertilised pasture	UK	< 10	Foster *et al.*, (1982)
Organic fertilisers only		< 25	
Inorganic fertilisers		50-70	
Arable	UK	30-120	Wilkinson & Greene (1982)
Grassland		12-30	
Rough grazing		3-6	
Grassland	Europe	4	OECD (1972)
Forest		0-6	
Arable row crops	Wisconsin, USA	2.8-26.9	Beaulac & Reckhow (1982)
Corn row crops	Iowa, USA	0.67-72.5	
Grassland	North Carolina, USA	2.4-18.1	

Table 3. Phosphorus export coefficients for rural land uses (after Mainstone *et al.*, 1996) (kg P ha^{-1} yr^{-1}) * median value; ** mean value

Land use	*Minimum*	*Average*	*Maximum*	*Reference*
Forestry	0.01		0.88	Loehr *et al.*, (1989)
	0.02	0.21*	0.83	Reckhow *et al.*, (1980)
Idle land	0.05		0.25	Loehr *et al.*, (1989)
Pasture	0.05		0.6	Loehr *et al.*, (1989)
	0.14	0.81*	4.90	Reckhow *et al.*, (1980)
Cropland	0.06		2.9	Loehr *et al.*, (1989)
	0.10		18.6	Reckhow *et al.*, (1980)
Mixed agriculture	0.08	0.91*	3.25	Reckhow *et al.*, (1980)
Natural export		0.1**		Chiaudani *et al.*, (1978)
	0.07	0.28*	0.65	Vighi and Chiaudani (1986)

CONCLUSIONS

The use of buffer zones to ameliorate contaminated run-off represents a curative approach to non-point source pollution and as such is not a long-term solution to pollution arising from agricultural land. However, buffer zones potentially offer a contributory mechanism for remediation until alternative, low-risk agricultural practices are adopted and in the longer term may provide protection against poor quality run-off should low-risk land management practices occasionally fail (Mainstone *et al.*, 1995).

The suggestion that the creation of economically viable buffer zones should be considered is based on the premise that the widespread adoption of a partially effective catchment protection measure will be of more benefit than a highly effective scheme which is limited in its level of take up. The acceptance of the principle of economically viable buffer zones by the farming community may open the door to schemes which are more effective in water quality terms, but which do not contribute to farm incomes.

There are a number of unknowns which require investigation before the efficacy of SRF for pollution control within buffer zones can be ascertained. Runoff from underdrained sites will bypass a buffer zone. However, SRF quickly invades drains, intercepting runoff but with possible negative repercussions for drainage upslope of the buffer. There is a considerable amount of root death associated with harvesting which along with the degradation of leaf litter may release large amounts of nutrients into soil drainage water. Forestry harvest systems are well-known to cause sediment runoff from the cut areas as a result both of the mechanical disturbance of soil by harvest machinery and the exposure of previously protected soil to the effects of rain impact. Similarly there may be unfavourable consequences from mechanical harvesting of SRF.

It is suggested that, despite these unknown, there is sufficient potential for SRF to form a component of a buffer zone and that this topic merits further investigation. In the first instance, this investigation could comprise the establishment of plantations to test the capacity of SRF to intercept both diffuse and point source pollution in different agricultural systems. If potential is demonstrated, the detailed investigation of the principle processes functioning could follow.

REFERENCES

Beaulac, M.N. and Reckhow, K.M. (1982) An examination of land use – nutrient export relationships. *Water Resource Bulletin,* **18**, 1013-1024.

Dillaha, T.A., Reneau, R.B., Mostaghimi, S. and Lee, D. (1989) Vegetative filter strips for agricultural non-point source control. *Transactions of the American Society of Agricultural Engineers,* **32**, 513-519.

Ericsson, T., Rytter, L. and Linder, S. (1992) Nutritional dynamics and requirements of short rotation forests. In: Mitchell, C.P., Ford-Robertson, J.B., Hinckley, T. and Sennerby-Forsse, L. (Eds.) *Ecophysiology of Short Rotation Forest Crops.* Elsevier Applied Science. pp 35-65.

Foster, S.S., Cripps, A.C. and Smith-Carrington, A. (1982) Nitrate leaching to groundwater. *Philosophical Transactions of the Royal Society, London* B296, 477-490.

Hasselgren, K. (1994) Kågeröd wastewater recycling experiment. In: Aronsson, P. and Perttu, K. (Eds.) *Willow vegetation filters for municipal wastewaters and sludges – a biological purification system.* Proc. of study tour and conference, Uppsala June 1994, Swedish University of Agricultural Sciences Report 50.

Hewitt, E.J. (1966) *Sand and Water Culture Methods used in the study of plant nutrition.* Technical Communication No. 222 (Revised) of the Commomwealth Bureau of Horticulture and Plant Crops. East Malling, Kent, Commomwealth Agricultural Bureaux, England.

HMSO (1992) *Department of Trade and Industry, Renewable Energy Advisory Group: Report to the President of the Board of Trade.* Energy Paper Number 60, November 1992, HMSO, London.

Johnes, P., Moss, B. and Phillips, G. (1994) *Lakes – Classification and Monitoring. A strategy for the classification of lakes.* NRA report 286/5/A.

Loehr, R.C., Ryding, S-O. and Sonzogni, W.C. (1989) Estimating the nutrient load to a water body. In: Ryding, S-O. and Rast, W. (Eds.) *The Control of Eutrophication of Lakes and Reservoirs.* Parthenon Publishing Group, Carnforth/UNESCO, Paris. 115-146.

Licht, L.A. (1994) *Populus spp. capabilities and relationships to landfill water management,* Air and Waste Management Ass. 87th Annual Meeting, Cincinnati, OH, June 1994.

Mainstone, C.P., Weddepol, J and Woodrow, D. (1995) *Land management techniques.* NRA report 453/5/Y.

Mainstone, C.P. and Schofield, K. (1996) Agricultural management for non-point pollution control, with particular reference to the UK. *European Water Pollution Control,* **6** (3), 21-30.

Mainstone, C.P., Davis, R.D., House, A. and Parr, W. (1996) *A review of methods for assessing and controlling non-point sources of phorphorus.* NRA report 562/5/W.

NOVEM (1992) Netherlands Agency for Energy and the Environment, *The Feasibility of Biomass Production for the Netherlands Energy Economy.* Project Number 71.140/0130.

OECD (1972) Water Management Sector Group, *Report of the Working Party on Fertilisers and Agricultural Waste Products,* Paris.

Reckhow, K.H., Khaleel, R., Overcash, M.R. and Westerman, P.W. (1980) *Modelling phosphorus loading and lake response under uncertainty: a manual and compilation of export coefficients.* USEPA report No. EPA 440/5-80-011.

Riddell-Black (1996) *Sewage Sludge Use in Intensive Coppice – Final Project Report.* WRc Report No. UC 2749.

Riddell-Black, D.M., Sims, R.E.H., Roygard, J., Clothier, B., Green, S. and Edwards, R. (1996) *Water and nutrient use by fuelwood species: preliminary results from an intensively monitored lysimeter study.* In: Proc. 9th European Bioenergy Conference, Copenhagen, June 1996 (in Press).

Sims, R.E.H., Mitchell, C.P. and Ford-Robertson, J.B. (1994) Disposal of effluent onto tree crops used for heat and power generation. In: Hall, D.O., Grassi, G. and Scheer, H. (Eds.) *Proc. Biomass for Energy and Industry,* 7th EC Conf., Florence, October 1992, 555-561.

Vighi, M. and Chiaudani, G. (1986) Una nova metodologia per la valutazione della capacit recettiva degli ambienti lacustri: il MEI e sua applicazione nei piani di risanamento, *Ingeneria Ambientale,* **15**, 239-246.

Wilkinson, W.B. and Greene, L.A. (1982) The water industry and the nitrogen cycle. *Philosophical Transactions of the Royal Society,* London B296, 459-475.

23 A critical review of the value of buffer zone environments as a pollution control tool

T.M. Addiscott

IACR-Rothamsted, Harpenden, Herts., AL5 2JQ, UK

Abstract

This paper takes the role of Devil's Advocate. The sceptical enquiry central to this role is directed towards a range of questions and issues as they occur to an outsider of the subject area. The paper summarises both the health and environmental concerns relating to nitrate and phosphate. It looks critically at the definitions of buffer zones and both their functions and apparent side-effects. Finally, it considers how effective buffer zones are in comparison with other approaches and whether they should be used in conjunction with other approaches.

INTRODUCTION

Agriculture and the aquatic environment have long been uneasy bed-fellows. The Domesday Book, compiled in 1086, takes note of the first known British drainage specialist, one Girard Fossarius, Gerard of the drain, who worked in the Somerset Levels. Sometime later, in 1252, the 'jurats' of Romney Marsh (its local government) are recorded as having had the power to repair the sea-wall and to control the ditches 'from time out of mind' (Purseglove, 1988). Girard and the jurats had an illustrious precursor; the Roman emperor Hadrian built a drain in the fen country as well as a wall in Northumberland.

Although it does not feature in the Domesday Book, the eutrophication of natural waters probably has an even longer history than land drainage. Ferguson *et al.* (1996) inferred from palaeolimnological studies that there has been a slow enrichment of Lake Windermere from the Iron Age onwards, as a result of the impacts of the Iron Age itself and subsequent forest clearance, pastoral farming, human settlement and quarrying.

Until recently, flooding had a much larger impact on the lives of most people than eutrophication. Much more was therefore done to restrict flows of water that interfered with agriculture than to restrict the outflow of nutrients from agriculture that interfered with life cycles in natural waters. This pattern has changed greatly in the past 25 years, with a rising perception that wetlands have a value in themselves and increasing concern about the polluting effects of the nutrients.

NUTRIENT PROBLEMS

Health concerns

Nitrate and phosphate have been the main sources of the concern about pollution. Public interest has centred mainly on the health risk believed to be posed by nitrate, and this has lead to the imposition of the EC's limit on nitrate in potable water, but there are indications that this risk has been greatly over-estimated. Cases of infantile methaemoglobinaemia seem to be associated with water from wells rather than from the mains water supply (e.g. Addiscott *et al.*, 1991; Addiscott, 1996), limiting the risk to a very small proportion of the population in the more developed countries (but not in the Third World). Furthermore, the supposed link between nitrate in water and stomach cancer has been shown to be non-existent in several epidemiological studies (e.g. Beresford, 1985; Forman *et al.*, 1985), and recent medical evidence suggests that nitrate is probably good for you. It forms the beginning of a

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0

chain of microbial and chemical reactions that form nitric oxide in the stomach and thereby destroy Salmonella and several other very undesirable bacteria in the gut (Dykhuizen *et al.*, 1996).

Environmental concerns

Although the evidence that nitrate is a risk to health seems very slim, nutrients washed from agricultural land definitely cause problems of eutrophication in surface waters (e.g. Ferguson *et al.*, 1996). Nitrogen and phosphate not only encourage the growth of crop plants, they also encourage that of water plants to an undesirable extent (The Royal Society, 1983; Ferguson *et al.*, 1996). Reeds can grow to excess, narrowing waterways and possibly overloading and damaging banks, while underwater plants foul fishing tackle and the propellers of boats. Water supply conduits can become clogged and machinery damaged. The greatest problem, however, is caused by algal blooms. These scum-like masses are unsightly on the surface of the water and they cause worse problems when they die, because the microbes that decompose them use oxygen needed by other species. These other species may decline in number or even die out, so that the food web and ecological balance of the stream or lake change, with the loss, in some cases, of the more desirable species of fish. The fact that some algal blooms include cyanobacterial species that are toxic to mammalian species, including humans and dogs (e.g. Bell and Codd, 1996) has added to public concern. This latter problem is not as new as might be believed. The record of cyanobacterial intoxication in animals dates back to at least 1853 (Bell and Codd, 1996).

To control algal blooms we need to know which nutrient, nitrogen or phosphate, has the dominant effect on their formation. Evidence reviewed recently by Ferguson *et al.* (1996) for lakes includes regression models that relate measures of phytoplankton, such as chlorophyll content, to the quantity or concentration of each nutrient in the lake. Relationships can be found for both nutrients, but that for phosphate is clearly the stronger and extends over five orders of magnitude in phosphate availability. The author's comment that this observation supports the view of most limnologists that phosphate has the greater influence of the capacity of the lake to sustain phytoplankton.

The overall conclusion about these 'problem nutrients' seems to be that nitrate is a threat to health only in the absence of a mains water supply, and that it causes problems in natural waters only when phosphate is present. Indeed, it could be argued that were it not for EC legislation there would be no nitrate problem. However, assuming that what is true of lakes is broadly true of rivers and streams, phosphate is a very definite threat to surface waters. This suggests that there may be different considerations for protecting underground aquifers and surface waters. Because of the EC limit for nitrate in potable water, the aquifers need primarily to be protected from nitrate. Phosphate is not a problem, because it is not perceived as a health risk. Also, so far as the UK is concerned, it will probably be sorbed by the chalk and limestone that form the matrix of many of our aquifers for at least as long as it takes us to sort these nutrient problems. On the other hand, phosphate is an immediate threat to surface waters, and where buffer zones are concerned with the protection of surface waters, restricting phosphate will often be the main objective. There can, however, be no firm distinction, because in many water systems there is continual interaction between surface water and aquifers (Burt, this volume; Goulding *et al.*, 1996) and in some areas most of the water for the mains supply is drawn from rivers.

BUFFER ZONES

What is a buffer zone?

A key part of any critical evaluation is to define what is being evaluated. The two key words seem to be 'buffer' and 'riparian'. Of the dictionary definitions relating to 'buffer', that of 'buffer-state' is the most apt: "a neutral state lying between two others, and serving to render less possible hostilities between them". More prosaically, 'riparian', which derives from ripa, the Latin word for a bank, means 'pertaining to or situated on the banks of a river'. Those researching the topic seem to use the terms ‘buffer’, ‘riparian’, ‘zones’ and ‘strips’ in various combinations, which include ‘buffer zones’ (Gaffney and Ross, 1995), ‘riparian zones’, (Gregory *et al.*, 1991), and ‘riparian buffer strips’ (Haycock and Pinay, 1993). Whether all these combinations mean the same thing seems uncertain.

Gaffney and Ross (1995) state that they use the term ‘buffer zone’ “to describe the edge of a field, adjoining the drainage ditches”, and they cite the MAFF recommendations for Environmentally Sensitive Areas in which farmers are advised to “avoid spreading fertiliser close to the edge of fields and into the ditches/hedges as this can cause pollution and damage the wildlife”. They add that “such practices would leave a 4-6 m unfertilised strip adjacent to ditches, thus providing a buffer zone for the movement of chemicals through the soil, an area in which the solutes would be: (i) adsorbed on to soil particles; (ii) chemically transformed; and/or (iii) taken up by vegetation”. Moorby and Cook (1992) were clearly thinking on the same scale as Gaffney and Ross (1995) when they investigated the effects of 5 m wide strips, defined unequivocally as ‘fertiliser-free grass strips’.

By contrast, Gregory *et al.* (1995) settled for a more general definition of ‘riparian zone’. “We define riparian zones functionally as three-dimensional zones of direct interaction between aquatic and terrestrial ecosystems. Boundaries of riparian zones extend outward to the limits of flooding and upward into the canopy of stream side vegetation. Dimensions of the zone of influence for a specific ecological process are defined by its unique spatial patterns and temporal dynamics.” There seems to be an important difference between this definition and those given in the previous paragraph. Gregory *et al.* (1995) give the impression that they are talking about a landscape feature, whereas Gaffney and Ross (1995) and Moorby and Cook (1992) are discussing modifications to farming practices at the edges of fields. There seems to be a difference in scale, which is emphasised by the comment of Lowrance *et al.* (1984) that: “In many parts of the south-eastern US coastal plain, land use on agricultural watersheds consists of row crops in well-drained uplands and native bottom land hardwood forests in stream side (riparian) areas”.

One solution to this terminological question would be to use the term ‘buffer strips’ for the agronomic modifications made at the edges of fields and to define ‘riparian zones’ at the scale of the landscape feature. This or some other convention may already have been adopted.

What are buffer zones meant to do?

In essence, buffer zones are intended to prevent the movement of nutrients from agricultural land into surface waters. The terms ‘filter’ and ‘sink’ are sometimes used to describe their function; both were used, for example, by Lowrance *et al.* (1984). A filter used in a laboratory removes solids from a solution but not usually the solutes; this, however, does not make the term inapplicable to buffer zones. Much of the phosphate that is carried by surface runoff is attached to eroded particulate material (Sharpley *et al.*, 1994), and this phosphate can be released into solution when the eroded material enters a stream or other surface water and experiences a change of pH or oxidative state. Bearing in mind the role of phosphate in the development of algal blooms, the removal by filtration of eroded phosphate-carrying material seems to be an important function of buffer zones. Filtration does not, in the strict sense, include the removal of nitrate, which is not sorbed by eroded material (except in soils that are far more acid than those likely to be considered in the present context). It could be said, however, that the vegetation in a buffer zone exerts a filtrative role in respect of nitrate and any phosphate remaining in solution.

The term ‘sink’ is used in thermodynamics and plant physiology as well as with respect to buffer zones. The former use implies irreversibility; heat cannot be reclaimed from a sink. In a plant, however, sinks rise and fall in dominance; the apical meristem becomes dominant early in the life of the plant but has to give way to the storage organ later. The buffer zone has elements of both these extremes. There are probably no irreversible processes in a buffer zone, but some nitrate and phosphate may end up in relatively inert organic matter in the soil. There are also changes in dominance; different processes rise and fall in importance with the seasons. During the main period of vegetative growth nitrate and phosphate are taken up actively by ground vegetation and trees.

Peterjohn and Correll (1984) found 'dramatic changes' in the water-borne nutrient load carried in surface runoff through about 50 m of riparian forest, and calculated that the trees had taken up about 77 kg ha^{-1} of nitrogen and 10 kg ha^{-1} of phosphorus during a year but had returned 62 and 7.8 kg ha^{-1} in leaf litter in autumn. Uptake of nutrients must have ceased once the leaves fell and transpiration ceased, so the trees must have ceased to be an effective sink at that time. Some of the nitrogen and phosphate in the leaves will have been translocated back into the tree when they senesced, and some will have become part of the leaf litter on the ground, from where it will eventually be released by soil microbes as nitrate or phosphate.

The sink described by Peterjohn and Correll (1984) is clearly not an irreversible sink, in that nutrients can be recycled from it, but its overall process is only slowly reversible, making it an effective sink for a period of time. But for how long will it remain effective? If the trees in the sink continue to absorb nitrate and phosphate from water flowing through it year after year, the time must surely come when the rate at which they are remineralised from the organic matter accumulated from the leaf litter is equal to the rate at which they are entering the system (e.g. Omernik *et al.,* 1981). In domestic parlance, the sink fills up and overflows. This kind of riparian zone is therefore a very useful way of 'holding the fort' while solutions are found to problems of nutrient pollution, but it does not seem to be the long-term answer. One solution to this problem is to harvest the trees at intervals to ensure an overall removal of nutrients from the zone (Lowrance *et al.,* 1984). This implies replanting and thence long-term planning, but this must surely be a worthwhile proposition, given the demand for wood. Failing this solution, the answer has ultimately to be found at the source of the nutrients.

Practically all vegetative nutrient sinks have the problem that they cease to function during the winter when nutrient losses from land are often at their greatest, so do buffer zones remain effective during winter? Haycock and Pinay (1993) addressed this question for nitrate in two riparian strips, one vegetated mainly with grass and the other dominated by poplar trees. They found that all the nitrate coming off the hillslope seemed to be removed in the first 5 metres of the riparian strip with poplars and about 84 percent of it in the grass strip. Their conclusion was that, although the vegetation played no active role in the removal of the nitrate, it contributed carbon substrates that facilitated the activities of nitrate-reducing bacteria. They did not investigate the fate of phosphate entering the strips but, unless the soil in the strips had been exposed to larger than usual amounts of it, the phosphate could well have been sorbed.

Are there side-effects of buffer zones?

Denitrification occurs in both saturated and unsaturated soils, usually more in the former. It was clearly very effective in the buffer zones investigated by Haycock and Pinay, presumably because they were saturated. We need to ask what else may have been happening in these zones, particularly if there was a gradual increase in anaerobicity. Patrick (1978), cited by Ross (1995), defined four likely stages in reduction processes:

1) Nitrate reduction begins before complete removal of oxygen. Manganic Mn reduced to manganous Mn at the same time.
2) Ferric Fe reduced to ferrous Fe when all oxygen and nitrate removed.
3) Sulphate reduced to sulphide (usually FeS or H_2S).
4) Carbon dioxide reduced to methane when most of the sulphate has been reduced.

Nitrate reduction is clearly desirable from a water quality viewpoint, but not from that of atmospheric quality, unless the reduction proceeds all the way to dinitrogen, N_2. The formation of the 'greenhouse' gas nitrous oxide, N_2O, is not something that should result from water quality measures. Should steps be taken therefore to intensify the reducing conditions? This seems questionable, because we do not want to encourage the reduction of sulphate to sulphide, particularly if H_2S may be formed, and we certainly do not wish to reduce carbon dioxide to methane, because the latter is another greenhouse gas. It would seem that buffer zones that depend on denitrification for their effectiveness need to be managed with great care.

BUFFER ZONES AND OTHER NUTRIENT CONTROL MEASURES

Buffer zones are not the only means of preventing nutrients from entering surface waters, and two questions arise:

1) How effective are buffer zones by comparison with other approaches?

2) Are buffer zones best used in conjunction with other approaches?

Comparison with other approaches

The other approaches that have been put forward include the following:

• Best (Conventional) Farm Practice

• Low input farming

• Organic farming

It is not possible to provide more than a very sketchy comparison between buffer zones and these approaches, because of the shortage of hard data on the restraints on nutrient losses provided by the farming systems and the buffer zones. The underlying problem is the shortage of data from 'average' farms, which are not especially well run and which do not adopt any of the approaches listed above and which can therefore be said to have 'average' nutrient losses. There also do not seem to be any comparisons between the three farming systems, and to complicate the issue further, there is the question discussed elsewhere (Addiscott *et al.*, 1991) as to whether, in comparisons between farming systems, the losses should be calculated on the basis of loss per hectare or loss per unit of commodity produced.

Best conventional farm practice can give quite small losses of nitrate from arable land, especially when winter wheat is grown. The Brimstone Farm Experiment has hydrologically isolated plots with field drainage overlying an impermeable clay subsoil, and its husbandry conforms to best farm practice. Losses from the plots growing winter wheat rarely exceed 40 kg ha^{-1} per year of nitrate-nitrogen, and are often less than 20 kg ha^{-1} (Catt *et al.*, 1992). It is difficult to assess whether this represents an improvement on the 'average' winter wheat crop because, as mentioned above, there are no data for the 'average' crop. However, Foster *et al.* (1982) estimated the loss of nitrate-nitrogen under a winter wheat crop from measurements in the underlying chalk, obtaining a figure of 70 kg ha^{-1}. There is no way of knowing whether these sets of figures are comparable, but they suggest that the improvements to be obtained from best farm practice may be of the order of 30 to 50 kg ha^{-1}. The restraint on nitrogen losses reported by Peterjohn and Correll (1984) was initially 77 kg ha^{-1}, but 62 kg ha^{-1} of nitrogen was returned to the soil in leaf litter and some will have been mineralised to nitrate. The safest conclusion seems to be that Best Farm Practice and riparian zones have restraining effects on nitrogen losses that are of the same order of magnitude, but neither is clearly more effective than the other. The restraints imposed by organic and low-input systems seem likely to fall within the same order of magnitude.

Use in conjunction with other approaches

There seems to be no question that a combination of (say) Best Farm Practice and a buffer zone will be more effective in protecting surface waters than either will be on its own. If organic farming is adopted with a view to decreasing nutrient losses, and particularly where animal manures are part of the system, the addition of buffer zones will make this option more effective.

There has been some discussion recently as to whether cuts in nutrient losses from agricultural land to the aquatic environment can be effected best by de-intensifying farming practice on the land currently used for farming, or by maintaining the present level of intensity, but on a smaller area of land, and using the remaining land for other purposes. One argument for the latter option is that there is a fixed environmental cost which is independent of the level of intensity. One component of this fixed cost is the effects of tillage, which tends to encourage mineralisation and thence, potentially, nitrate leaching. Tillage also uses fossil fuels and adds to the emission of carbon dioxide and nitrogen oxides. Bertilsson (1992) made a study of farming that considered emissions of gases to the atmosphere and nutrients to natural waters for the whole of Sweden, taking account of off-farm processes such as the production and transport of fertiliser. He showed that the achievement of a predetermined level of agricultural production without nitrogen fertiliser led to greater emissions of gases and nutrients than the achievement of the same production with the fertiliser. The difference lay simply in the smaller amount of land that had to be cultivated when the fertiliser was used. Bertilsson assumed initially that the land not cultivated would be released for 'nature', but he also calculated the effects of using the spare land for the production of willows as an energy crop. This gave the more intensive system an even greater environmental advantage, because of the removal of carbon dioxide from the atmosphere.

Much is to be gained by thinking at the scale of the landscape and integrating Bertilsson's approach with that of the American riparian researchers Peterjohn and Correll (1984), Lowrance *et al.* (1984) and Gregory *et al.* (1991). The spare land left by intensive farming in Bertilsson's approach could be assigned to substantial riparian zones. Deciduous and other 'amenity' trees could be established if not already there and willows could harvest energy while removing carbon dioxide from the atmosphere and nutrients from water flowing through the zone. An approach of this nature has, no doubt, been suggested before, but it implies a much greater degree of long-term landscape planning than has been achieved in the past.

Animal production systems

Most of the papers cited in this brief review have been concerned with protecting natural waters from arable agriculture. Such waters need to be protected from animal production systems too. Cattle are often to be seen grazing near rivers, and their excreta can easily be washed in by surface runoff. Slurry or manure applied to grassland present a similar threat, and buffer zones ought to be effective as filters both for physically restraining solid material and for supporting vegetation that removes nitrate and phosphate from the water passing through the zones. We also need to consider the adjuncts to grassland as well as the grassland itself. Farmyards and other stock-handling areas are a common source of nutrients that pollute waterways, because they tend to have concrete floors from which excreta are rapidly washed. Slurry tanks and silage clamps can also leak effluent into nearby water. Buffer zones may be able to ameliorate all these problems if strategically placed.

SOME QUESTIONS

This has been very much an outsider's look at buffer zones and the author makes no claim to any expertise in the topic. The review has raised the following questions, some of which may already have been addressed.

- Does the terminology of the topic perhaps need to be clarified? For example, do the terms 'buffer' and 'riparian' carry the same meaning? And surely 'strips' and 'zones' are not really synonymous?
- There seems to be good evidence that buffer strips and riparian forest zones can intercept nutrients and so prevent them entering surface waters. Is it known whether they can become 'saturated' with nutrients and, if so, over what period of time is this likely to occur?
- Buffer zones can clearly operate at various scales. Have the efficiencies with which they function at different scales been investigated?
- Have buffer zones for animal production systems received the same attention as those for arable agriculture?

- Has any kind of risk analysis been made that assesses and compares the problems caused by nitrate in water and nitrous oxide in the atmosphere? Also, is it known whether buffer zones that rely on denitrification produce significant quantities of methane or H_2S? This ought to be part of the risk assessment.
- Is the behaviour of buffer zones sufficiently well-understood to enable risk management strategies to be put in place?
- Are there possibilities for getting 'added value' from buffer zones, in the form of energy and wood crops, for example? If so, are they being followed up?
- Most evidence suggests that it is phosphate rather than nitrate that determines whether algal blooms form in freshwaters. Nitrate is now said to be good for you. Is research on buffer zones geared towards the control of nitrate or phosphate and is a change needed?
- Should we be thinking about a radical reorganisation of the landscape in the long-term?

ACKNOWLEDGEMENTS

The author is grateful to Nick Haycock, Tim Burt and Keith Goulding for the invitation to contribute to the conference and to Arthur Gold for providing copies of some key articles on buffer zones.

REFERENCES

Addiscott, T.M. (1996) Fertilizers and nitrate leaching. *Issues in Environmental Science and Technology,* **5**, 1-26.

Addiscott, T.M., Whitmore, A.P. and Powlson, D.S. (1991) *Farming, fertilizers and the nitrate problem.* Wallingford: CAB International.

Bell, S.G. and Codd, G.A. (1996) Detection, analysis and risk assessment of cyanobacterial toxins. *Issues in Environmental Science and Technology,* **5**, 109-122.

Beresford, S.A. (1985) Is nitrate in drinking water associated with gastric cancer in the urban UK? *International Journal of Epidemiology,* **14**, 57-63.

Bertilsson, G. (1992) Environmental consequences of different farming systems using good agricultural practices. *Proceedings of the International Conference of the Fertilizer Society, Cambridge, 16-17 December 1992.*

Catt, J.A., Christian, D.G., Goss, M.J., Harris, G.L. and Howse, K.R. (1992) Strategies to reduce nitrate leaching by crop rotation, minimal cultivation and straw incorporation in the Brimstone Farm Experiment, Oxfordshire. *Aspects of Applied Biology,* **30**, 255-262.

Dykhuizen, R.S., Frazer, R., Duncan, C., Smith, C.C., Golden, M., Benjamin, N. and Leifert, C. (1996) Antimicrobial effect of acidified nitrite on gut pathogens: Importance of dietary nitrate in host defence. *Antimicrobial Agents and Chemotherapy,* **40**, 1422-1425.

Ferguson, A.J.D., Pearson, M.J. and Reynolds, C.S. (1996) Eutrophication of natural waters and toxic algal blooms. *Issues in Environmental Science and Technology,* **5**, 27-41.

Forman, D., Al-Dabbagh, A. and Doll, R. (1985) Nitrate, nitrite and gastric cancer in Great Britain. *Nature,* **313**, 620-625.

Foster, S.S.D., Cripps, A.C. and Smith-Carrington, A. (1982) Nitrate leaching to groundwater. *Philosophical Transactions of the Royal Society,* **B296**, 477-489.

Gaffney, S.W. and Ross, S.M. (1995) Field edge solute processes in fen peats of the Somerset moors. In: Hughes, J.M.R. and Heathwaite, A.L. (Eds.) *Hydrology and hydrochemistry of British wetlands.* Chichester: John Wiley and Sons. pp. 199-221.

Goulding, K.W.T., Matchett, L.S., Heckrath, G., Webster, C.P., Brookes, P.C. and Burt, T.P. (1996) Nitrogen and phosphorus flows from agricultural hillslopes. In: Anderson, M.G. and Brooks, S.M. (Eds.) *Advances in hillslope processes.* Chichester: John Wiley and Sons. Ch 9, pp. 213-227.

Gregory, S.V., Swanson, F.J., McKee, W.A. and Cummins, K.W. (1991) An ecosystem perspective of riparian zones. *BioScience,* **41**, 540-551.

Haycock, N.E. and Pinay, G. (1993) Groundwater nitrate dynamics in grass and poplar vegetated riparian buffer strips during the winter. *Journal of Environmental Quality,* **22**, 273-278.

Lowrance, R., Todd, R., Fail, J., Hendrickson, O., Leonard, R. and Asmussen, L. (1984) Riparian forests as nutrient filters in agricultural watersheds. *BioScience,* **34**, 374-377.

Moorby, H. and Cook, H.F. (1992) The use of fertilizer-free grass strips to protect dyke water from nitrate pollution. *Aspects of Applied Biology,* **30**, 231-234.

Omernik, J.M., Abernathy, A.R. and Male, L.M. (1981) Stream nutrient levels and proximity of agricultural and forest land to streams: Some relationships. *Journal of Soil and Water Conservation,* **36**, 227-231.

Patrick, W.H. (1978) Critique of Measurement and Prediction of Anaerobiosis in Soils. In: Nielson, D.R. and MacDonald, J.G. (Eds.) *Nitrogen in the Environment. I. Nitrogen Behaviour in Field Soil.* London: Academic Press. pp. 449-457.

Peterjohn, W.T. and Correll, D.L. (1984) Nutrient dynamics in an agricultural watershed: observations on the role of a riparian forest. *Ecology,* **65**, 1466-1475.

Purseglove, J. (1988) *Taming the flood. A history and natural history of rivers and wetlands.* Oxford: Oxford University Press.

Ross, S.M. (1995) Overview of the hydrochemistry and solute processes in British wetlands. In: Hughes, J.M.R. and Heathwaite, A.L. (Eds.) *Hydrology and hydrochemistry of British wetlands.* Chichester: John Wiley and Sons. pp. 133-181.

The Royal Society (1983) *The Nitrogen Cycle of the United Kingdom: A Study Group Report.* London: The Royal Society.

Sharpley, A.N., Chapra, S.C., Wedepohl, R., Sims, J.T., Daniel, T.C. and Reddy, K.R. (1994) Managing agricultural phosphorus for protection of surface waters. *Journal of Environmental Quality,* **23**, 437-451.

24

Long and short roads to riparian zone restoration: nitrate removal efficiency

M.T. Downes[1], C. Howard-Williams[1] and L.A. Schipper[2]

[1] *NIWA, PO Box 8602, Christchurch, New Zealand*
[2] *Landcare Research NZ Ltd, Private bag 3127, Hamilton, New Zealand*

Abstract

Many riparian zones have been modified or destroyed by anthropogenic practices and there is a need to develop techniques which restore their ability to protect surface water quality. Because of the multiple functions of riparian zones, there needs to be a clear understanding of the objectives of restoration and of the mechanisms by which riparian zones function. In this context, we discuss three approaches we have taken in riparian restoration which differ in their objectives and timescale of restoration. Nitrate was used as the indicator of water quality changes.

In the first two approaches, two types of grazed pasture streams were fenced off to protect stream banks from stock damage: 1) Streams with a strong, spring-fed source in which riparian vegetation modified the water quality in the stream channel; 2) Streams fed by surface flows from lateral groundwater in which riparian vegetation modified the water before it reached the stream channel. In stream type 1, water quality was improved in the first few years by nutrient uptake by non-woody instream vegetation. This declined with time as woody plants developed. In stream type 2, nutrient uptake was microbially mediated along the edges of the channel.

The third approach considered a stream receiving sub-surface flow from lateral groundwater. In this case a trench was dug, a metre deep into groundwater and parallel to the stream. The excavated soil was mixed with sawdust (30% v/v) and returned into the trench. Sawdust provided a carbon source for groundwater nitrate removal by denitrification. Denitrifying enzyme activity in the trench increased at least 10-fold.

These studies have demonstrated a range of timescales for restoration. Riparian restoration from pasture to a cover dominated by native vegetation (as opposed to introduced pasture plants) takes about 30 years. However, the third approach shows that modifying the path of groundwater flow has the potential to quickly decrease groundwater inputs of nitrate to surface waters within a matter of months.

INTRODUCTION

Wetlands are among the most threatened ecosystems in the world and finding practical approaches for their restoration is an expanding area of research (Mitsch, 1992; Petersen, 1992). A feature which distinguishes riparian ecosystems from many other wetlands is that they are spatially linear to surface waters. Water and nutrients are transported through riparian zones from upland terrestrial soils to surface waters by surface and subsurface flows (Burt, this volume). As such, they are ideally placed to intercept non-point source pollutants originating from terrestrial ecosystems, prior to entering surface waters (Gilliam, 1994). While restoration of riparian zones may be effective at improving water quality, they should not be seen as a solution to all problems. They represent a last line of defence for surface water quality protection and are not a substitute for good upslope land management practices.

When considering restoration of riparian zones, it is important to clearly identify the primary objective of restoration. Riparian zones have other important ecological values besides water quality protection, such as protection of surface water habitats, habitat for riparian species and aesthetic value (Sedell *et al.*, 1991). In some cases, riparian restoration for water quality purposes may also meet these other

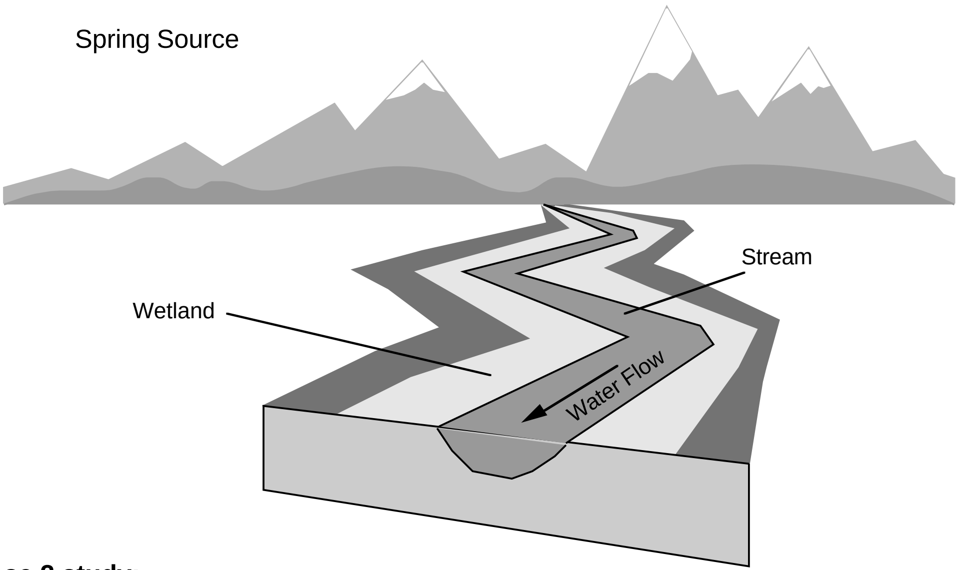
Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0 © 1997 Quest Environmental.

objectives. Water quality protection provided by riparian zones is dependent on differing physical and biological properties of the riparian zone. On the one hand, the retention of sediment and phosphorus is dependent on surface roughness of a riparian zone and it has been suggested that this is best provided by low thick vegetation such as grasses rather than a closed canopy forest (Osborne and Kovacic, 1993). On the other hand, greater rates of nitrate removal have been observed in forest systems or organic riparian wetlands than in grassed riparian zones (Cooper, 1990; Haycock and Pinay, 1993). This has been attributed to the requirement by denitrification of a source of organic matter, both for energy and to promote anaerobic conditions. Grassed and forested riparian zones provide very different ecosystems for plants and animals. Recommendations have been made for the restoration of riparian zones to consist of multiple zones of differing habitat (Lowrance *et al.*, 1995).

Figure 1. Three case studies: Case 1) a strong spring source with little lateral flow, Case 2) lateral springs with overland flow to the stream channel and Case 3) lateral springs with groundwater flow to the stream channel.

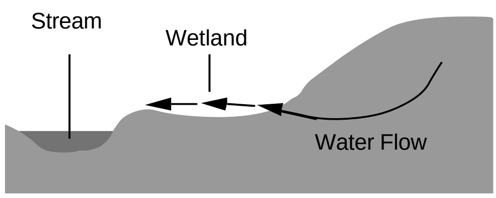
Case 1 study:

Spring Source



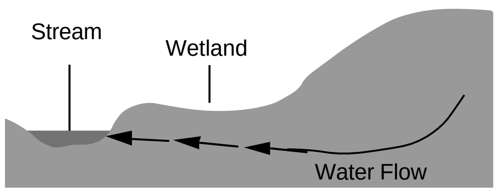
Case 2 study:

Lateral Springs / overland flow



Case 3 study:

Lateral Springs / groundwater flow



Wide differences in catchment hydrology are, in part, responsible for the difficulties in deriving a prescriptive approach for riparian zone restoration. Broadly speaking, pollutants can enter surface waters by overland flow, subsurface flow or via springs, depending on the hydraulic properties of the soils, geology, evapotranspiration and rainfall of the catchment. Planting of riparian zones can also alter the catchment hydrology (Smith, 1992). As a result, the degree of contact between the transported pollutants and the riparian soils varies both spatially and temporally. Relatively broad riparian zones may be needed to trap sediment and phosphorus in overland flow, while relatively narrow wetland seeps can remove large amounts of nitrate in groundwater inputs, provided that the flow rates are not too great. Riparian zones can also remove nutrients directly from stream waters (Howard-Williams *et al.*, 1986; Triska *et al.*, 1993).

While recommendations for restoration have been made, there are still many questions outstanding, which have been asked by both researchers and land managers (Gilliam, 1994). What width should they be? What vegetation should be planted? Should they be placed along all streams? How long before they are effective? How long will they remain effective? We have few scientifically-based answers to these questions as there have been few long-term controlled studies. Furthermore, there is unlikely to be a single prescription for restoration of riparian zones since there are wide differences in soil types, climate, hydrology and objectives for restoration. Once an objective for a particular restoration project has been decided, a clear understanding of the renovation functions of riparian zones is needed.

Here we present three studies of riparian zones (Fig. 1), and through them highlight some of the difficulties of riparian restoration. The first is a long-term study of a stream that runs through agricultural land and which has had riparian zones retired for 20 years. It focuses on changes to the main stream channel. The second and third studies describe preliminary results of modifications to water moving across riparian zones before reaching the channel. In case study two, modifications occurred to surface water flow and in case study three, to groundwater flow.

CASE STUDY 1

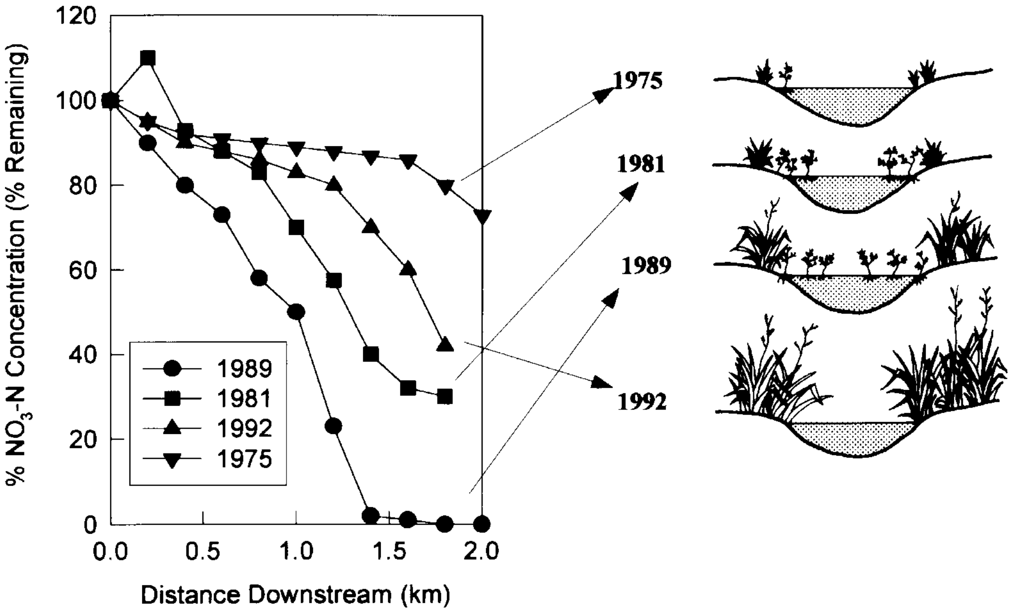
Riparian fencing of the streams flowing through pasture land in the Lake Taupo catchment was systematically conducted in the 1970s as a soil conservation measure and to improve water quality (Howard-Williams *et al.*, 1983).

We have studied the restoration process of one of these streams over the last 20 years. Headwater springs provide the source of water for the stream (Whangamata Stream, 38°37'S, 175°55'E), which has a base flow discharge of between 0.04 and 0.1 $m^3 s^{-1}$. Flood flows account for only 5% of total flow over a year. The low variance in the discharge means that the stream is ideal for the study of nutrient dynamics and water quality. Regular measurements were made at three sampling sites. The 'top' site was 2.0 km from Lake Taupo and the 'bottom' site was close to where the stream discharged into the lake. Flows were estimated at a site approximately 1.0 km from the lake using an acoustic stage recorder set in a calibrated section flume. There were no tributary inflows between sites.

Duplicate water samples were collected at each site at weekly intervals from 1979 to 1981 and at fortnightly intervals between 1982 and 1993. Vegetation surveys were carried out by walking the length of the stream and constructing species lists for plants growing in the stream channel, on the banks and on wet soil on either side of the banks (Howard-Williams *et al.*, 1987 and unpublished data).

More detailed descriptions of field and laboratory methods are given in Downes (1978 a, b) and Howard-Williams *et al.* (1982).

Figure 2. Downstream changes in nitrate concentration during the summers of 1975, 1981, 1989 and 1992 in the Whangamata Stream together with a diagrammatic representation of changes in the streambank vegetation.

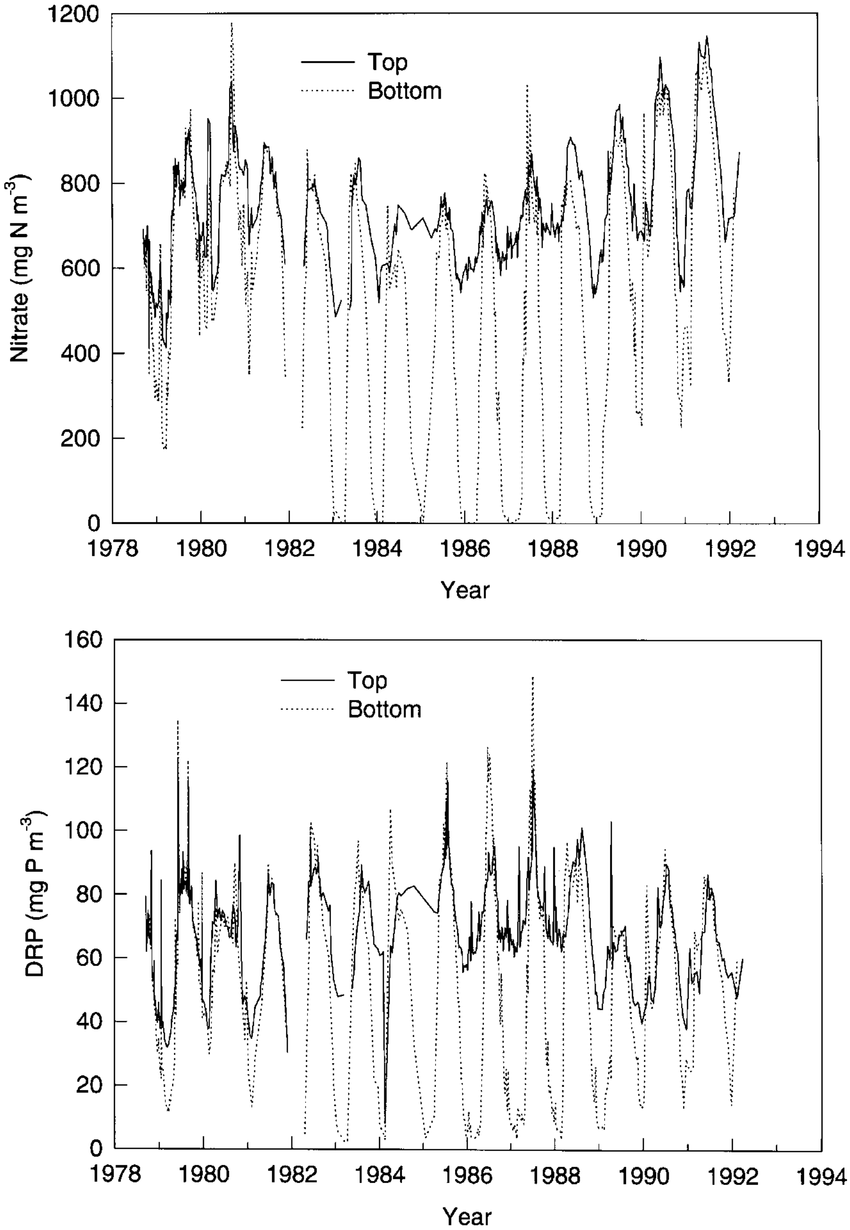


Following riparian protection, the stream was left alone. A vegetation succession began initially with the development of watercress *(Rorippa nasturtium-aquaticum)* and sweetgrass (*Gyceria* spp.), and was followed by the gradual invasion of tall wetland vegetation species such as flax *(Phormium tenax)*, cabbage tree *(Cordyline australis)* and finally, woody species and treeferns (Howard-Williams and Pickmere, 1994). Nitrate and dissolved reactive phosphorus (DRP) were progressively removed downstream (Fig. 2) and, following natural riparian succession, this removal capacity increased considerably over the first decade (Fig. 2). This coincided with the rapid development of an aquatic and semi-aquatic in-stream flora which also stripped out the nutrients (Howard-Williams *et al.*, 1983; Howard-Williams and Downes, 1984). The difference in upstream and downstream concentrations in a 2 km reach (Fig. 3) between 1979 and 1988 demonstrate this well. Mass flow calculations, from concentration and stream discharge data, demonstrated that the amounts of nutrient removed each year increased for the first 10 years and then decreased (Table 1). The pattern of increase and then decrease was consistent with the successional development of a tall wetland vegetation and finally invasion of woody species which shaded out the herbaceous, nutrient removing, in-stream and bank vegetation. In this case, the primary mechanism of nitrate removal was plant uptake. Plant tissue nitrogen is a transient pool and a question arises regarding the fate of the nitrogen following tissue decomposition.

Table 1. Biological removal (kg yr^{-1}) of nitrate nitrogen and dissolved reactive phosphorus from the Whangamata Stream water between 1986 and 1993.

	Mass removed (kg yr^{-1})	
Year	*Nitrate-N*	*DRP*
1986-87	475	47.4
1987-88	787	71.7
1988-89	558	48.0
1989-90	413	33.8
1990-91	239	14.6
1991-92	234	20.7
1992-93	125	10.7

Figure 3. Concentrations of a) nitrate and b) DRP at the top and bottom sites of the 2 km study reach of the Whangamata Stream from 1978 to 1992.



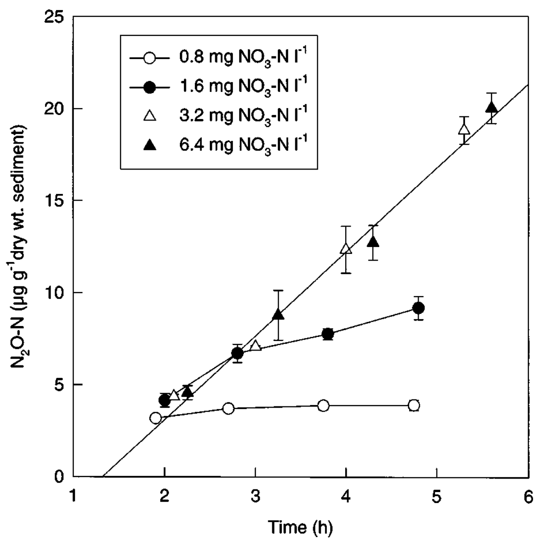
Studies on the decomposition of in-stream vegetation during the period of maximum nutrient stripping showed that about 50% of the nitrate removed by the channel and bank vegetation was transformed and flowed downstream as particulate nitrogen and dissolved organic nitrogen and 50% was lost from the system by denitrification. Denitrification enzyme activity (DEA) was measured on anoxic sediment samples by acetylene inhibition of nitrous oxide reductase (Yoshinari and Knowles, 1976; Balderstone *et al.*, 1976) and also *in situ* by placing hoods over the streambank sediments where the plants were decomposing, and adding acetylene to the headspace. In the latter studies, we measured natural N_2O release and N_2O release following addition of acetylene to the substrate under the hoods. Details are given in Howard-Williams and Downes (1984).

Table 2. *In situ* denitrification rates in the Whangamata Stream. Data show means from six soil chambers located across the stream on different dates. (SE = standard error).

	Denitrification rate (mg N m⁻² day⁻¹)	
Date	*Mean*	*S.E.*
25/5/80	54.2	27.20
11/6/80	40.6	21.40
2/7/80	50.9	20.26
6/8/80	83.5	55.91
3/9/80	60.9	24.66
9/10/80	1.2	0.62
19/11/80	4.6	4.49
12/2/81	20.1	14.05
31/3/81	30.1	10.59

The data showed a very patchy distribution of denitrification with mean *in situ* rates ranging from 1.2 to 83.5 mg N m^{-2} day^{-1} (Table 2). Rates were maximal in winter, between May and September, when water levels were lowest. These rates are lower than the rate of nitrate uptake by plants (94 – 11400 mg N m^{-2} day^{-1}, Howard-Williams *et al.*, 1982). In addition, nitrification rates measured in the decomposing plant material (Howard-Williams *et al.*, 1983) were also much higher than the denitrification rates (238 mg N m^{-2} day^{-1}) which indicates that in this case nitrate was not limiting denitrification. A set of laboratory DEA assays on streambank sediment for different levels of nitrate addition are shown in Figure 4. Rates were initially independent of nitrate concentrations but rate limitation was evident for initial nitrate concentrations of 0.8 and 1.6 mg N l^{-1}. When nitrous oxide production stopped in these assays, after 2 and 3 hours respectively, about 40% of the added nitrate had been reduced to nitrous oxide. At the higher nitrate additions, nitrous oxide production continued for more than six hours indicating that denitrification was not limited by substrates other than nitrate over this timescale. Nitrous oxide production also stopped in these higher nitrate addition assays when 40% of the added nitrate had been reduced to nitrous oxide. This suggests competitive inhibition of denitrification, possibly by dissimilation of nitrate to ammonia, which is favoured by habitats where carbon is abundant (Tiedje *et al.,* 1982). In such a system, with luxuriant, herbaceous stream vegetation developing along the channel, enhancement of nitrate removal by denitrification would not depend upon carbon additions. Development of dense riparian vegetation with high carbon turnover rates would be an advantage in a system where nitrogen removal was dependent on denitrification.

Figure 4. Denitrification enzyme activity assay on Whangamata Stream sediments showing the accumulation of N_2O-N with time after additions of different amounts of NO_3-N.
The NO_3-N values are the concentrations in the sediment slurry immediately after addition.



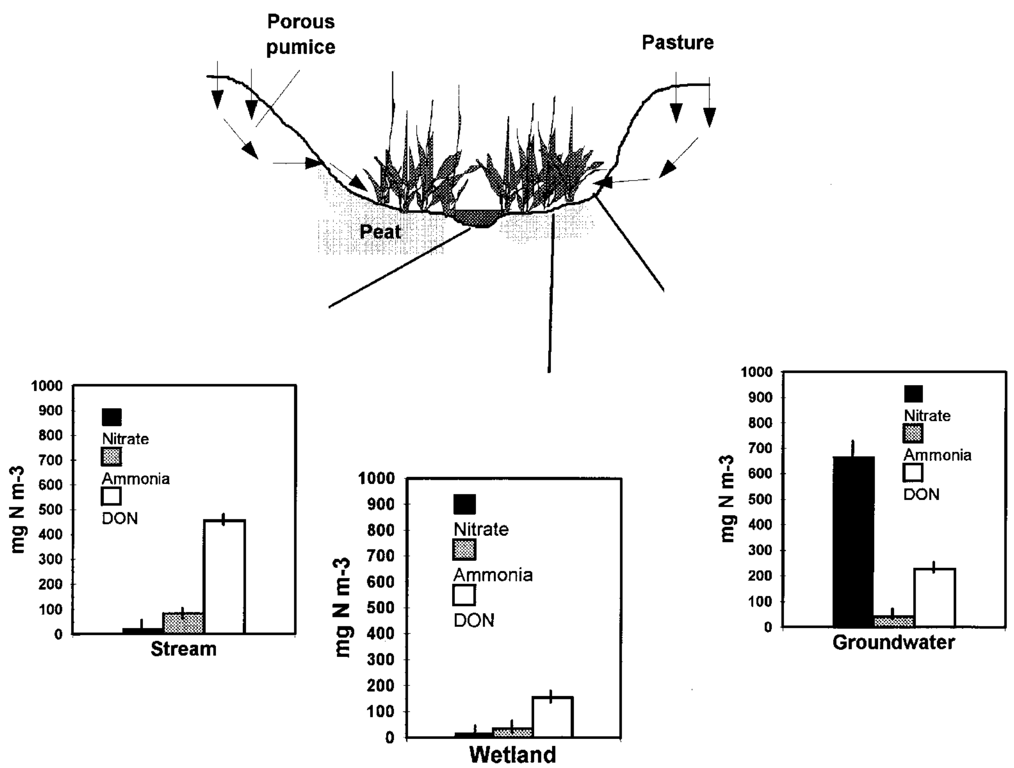
CASE STUDY 2

This was a study of nitrate removal from lateral surface waters moving through a protected riparian zone along Tutaeaua Stream (38°37'S, 175°50'E) in the Lake Taupo catchment. In this stream, the channel was shaded by fully-developed tall wetland vegetation and the nitrate-rich water from the surrounding pasture lands seeped into the riparian wetland (Fig. 5). Most of this water flowed across the surface of a dense, organic, wetland soil before reaching the main stream channel. The main stream channel, densely overgrown with flax *(Phormium tenax),* had low nitrate concentrations throughout its length. DEA was measured on subsamples from three soil cores collected in the riparian wetland. The subsamples were taken from 5 depths in each core (Table 3). The data showed that DEA was fairly uniform in the top 10.5 cm of sediment (Table 3). On an area basis, the top 1 cm of sediment has a denitrification capacity of 9.07 mg N m^{-2} day^{-1}. The assays also indicated that denitrification was more efficient in the surface sediments with up to 80% of nitrate denitrified. These high rates of DEA could potentially be reached in the Case Study 1 stream when the vegetation succession progresses further.

Table 3. DEA in soil cores from the Tutaeaua Stream Wetland.

Soil Depth (cm)	*Mean DEA (μg N g^{-1} h^{-1})*	*S.E.*
0-1.0	1.505	±0.5
1.0-2.5	0.925	±0.3
2.5-4.5	1.135	±0.05
4.5-6.5	1.465	±0.7
6.5-10.5	0.675	±0.4

Figure 5. Changes in dissolved nitrogen from the lateral groundwater inflow, through the wetland to the Tutaeaua Stream. Each histogram is the mean of 3 sites and 30 weekly samples for each water type. Error bars = ± 1 S.E. (Original data from Howard-Williams, Pickmere and Davies, 1986)



Mass flow calculations, derived from channelled flows across the wetland, indicated that most of this nitrate was denitrified during passage through this lateral wetland. In-stream processes played no observable role in the main stream channel.

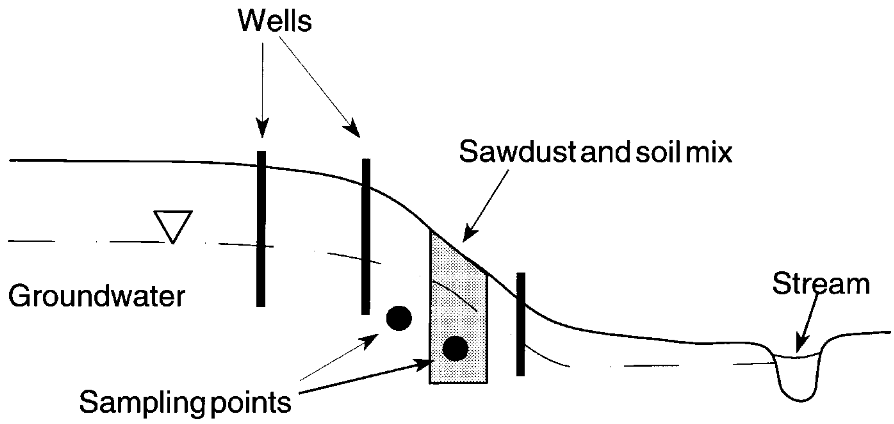
CASE STUDY 3

In the third case study we have taken advantage of the lateral movement of groundwater to artificially enhance the denitrification process.

Intensive agriculture, land application of waste and other land uses can result in nitrate leaching to groundwater and then polluting surface waters. In these cases, a motivating force behind the restoration of riparian zones is the removal of nitrate from groundwater prior to discharge to surface waters.

This case study was conducted at a Central North Island dairy farm (37°57′S, 175°27′E) which had nitrate concentrations in shallow groundwater of up to 13 mg L^{-1}. The groundwater discharged into a small stream which ran through the property. The soils were poorly drained rhyolitic alluvium, deposited on a terrace ridge. A site was selected adjacent to the stream below a pasture grazed by dairy cows and spray-irrigated with dairy factory effluent. A trench was dug into the slope of the terrace parallel to the stream and was 1.5 m deep, 35 m long and 1 m wide (Fig. 6). The soil removed from the trench was mixed with 40 m^3 of untreated sawdust and then returned to the trench. On the slope of the terrace, groundwater rose to within 20 cm of the surface in winter but was deeper (1.2-1.5 m) in summer when rainfall was lower.

Figure 6. Cross section of the artificial riparian zone showing sampling sites in the groundwater and in the soil/sawdust mix.



A month after construction a number of samples were taken for biochemical analysis. Soil samples were taken using a Dutch auger from groundwater soils above the trench and from the soil/sawdust mix in the trench below the groundwater (Fig. 6). Microbial biomass was determined with the method of Vance *et al.* (1987) using a k_{ec} factor of 0.41. Basal respiration was determined by measuring carbon dioxide production during a seven-day aerobic incubation. Denitrifying enzyme activity (DEA) was determined by the method of Tiedje *et al.* (1989). *In situ* denitrification was measured by determining nitrous oxide production during a 24-hour bottle incubation with 10% (v/v) added acetylene in the headspace. Samples collected from below the groundwater were flushed with nitrogen prior to incubation to mimic the low oxygen concentrations found in saturated soils. Nitrate concentration was determined by standard auto-analyser techniques following KCl (2M) extraction of the samples (Blakemore *et al.*, 1987). Total carbon was determined using a LECO combustion furnace (Blakemore *et al.*, 1987). A further experiment was conducted to determine the factors limiting denitrification.

Samples of the soil/sawdust mix were incubated anaerobically in a shaking incubator at 25°C with added acetylene (10% v/v) and chloramphenicol. Samples were amended with: 1) glucose and nitrate (i.e. DEA measurement); 2) glucose only; and 3) nitrate only. The rate of nitrous oxide production was determined during a one-hour incubation.

The addition of sawdust to soil increased the total carbon content from 0.16% in the groundwater soils to 5.2%. The increased organic matter content of the groundwater was matched by a general increase in microbial biomass and activity in comparison to adjacent groundwater soils (Table 4). Denitrifying enzyme activity also increased in the soil/sawdust mix but was less than found in other riparian zones (Cooper, 1990; Groffman *et al.*, 1992; Schipper *et al.*, 1993). Nitrate concentrations in the trench were less than half that measured directly upslope, indicating that nitrate removal was occurring in the trench. Despite apparent nitrate removal, *in situ* denitrification rates were below detection limits for both sampling locations and it was thought that this was due to the very low nitrate content in the trench. Nitrate concentrations were well below the apparent k_m values for denitrification (Myrold and Tiedje, 1985).

Table 4. Biochemical properties of samples taken from groundwater soils and from trench soil/sawdust mix.

Property	*Units*	*Groundwater soils*	*Trench soil / sawdust mix*
		Mean (S.E.)	
Nitrate	$\mu g\ N\ g^{-1}$	1.15 (0.32)	0.52 (0.09)
Basal respiration	$\mu g\ C\ g^{-1}\ h^{-1}$	0.16 (0.05)	5.11 (0.23)
Microbial Biomass	$\mu g\ N\ g^{-1}$	62 (16)	450 (40)
Denitrifying enzyme activity	$\mu g\ N\ g^{-1}\ h^{-1}$	8.1 (3)	122 (42)

Average nitrous oxide production rates for glucose and nitrate amendment were 65 $\mu g\ N\ g^{-1}\ h^{-1}$, for nitrate amendment 66 $\mu g\ N\ g^{-1}\ h^{-1}$, and for glucose amendment 2 $\mu g\ N\ g^{-1}\ h^{-1}$. This confirmed that denitrification in the trench was nitrate limited, probably due to the decreased nitrate leaching during summer when the denitrification rate was measured. It is anticipated that groundwater nitrate concentrations will increase during winter, as previously measured, and that denitrification will be stimulated.

The key limitation of this approach for restoring riparian nitrate removal is the long-term sustainability of the added carbon substrate. We added 40 m^3 of sawdust or approximately 16 tonnes (49% C). At the current rate of microbial respiration the added C will be consumed in less than three years. The rate of respiration is likely to decrease as the more degradable components of the sawdust are consumed. Whether the declining organic matter quantity and quality will provide suitable carbon sources for denitrifying bacteria will require further monitoring. A further consideration for nitrate removal is the groundwater flow rate. Warwick and Hill (1988) measured high rates of denitrification in a small riparian zone but nitrate removal was limited by the short residence time (<1 h) of the nitrate as it passed through the zone. Since the trench is only 1.5 m wide, a high rate of groundwater flow may result in incomplete nitrate removal due to a short retention time. The constructed riparian zone is a potential short-term solution but may be a useful approach if used in conjunction with fencing and planting of riparian areas. The added organic matter may sustain nitrate removal until established riparian vegetation supported nitrate removal through above- and below-ground organic matter inputs.

CONCLUSION

Many studies have concluded that riparian wetlands are useful for surface water quality protection and call for the restoration of these ecosystems. Yet there are few reported controlled studies of riparian restoration, successful or otherwise. Conducted studies have generally been short-term but riparian restoration must be viewed in the longer term. This should not stop us estimating the appropriate riparian zones for specific soils and climates and taking a research by management approach (Walters and Holling, 1990). The key to successful development of appropriate riparian restoration is the early identification of the primary objectives for restoration and a clear understanding of the mechanisms by which riparian zones protect surface water quality.

REFERENCES

Balderston, W.L., Sherr, B. and Payne, W.J. (1976) Blockage by acetylene of nitrous oxide reduction in *Pseudomonas perfectomarinus. Applied and Environmental Microbiology,* **31**, 504-508.

Blakemore, L.C., Searle, P.L. and Daly, B.K. (1987) *Methods for chemical analysis of soils.* NZ Soil Bureau Scientific Report, 80.

Cooper, A.B. (1990) Nitrate depletion in the riparian zone and stream channel of a small headwater catchment. *Hydrobiologia,* **202**, 13-26.

Downes, M.T. (1978 a) An improved hydrazine reduction method for the automated determination of low nitrate levels in freshwater. *Water Research,* **12**, 673-675.

Downes, M.T. (1978 b) An automated determination of low reactive phosphorus concentrations in natural waters in the presence of arsenic, silicon and mercuric chloride. *Water Research,* **12**, 743-745.

Gilliam, J.W. (1994) Riparian wetlands and water quality. *Journal of Environmental Quality,* **23**, 896-900.

Groffman, P.M., Axelrod, E.A., Lemunyon, J.L. and Sullivan, W.M. (1992) Denitrification in grass and forest vegetated filter strips. *Journal of Environmental Quality,* **20**, 671-674.

Haycock, N.E. and Pinay, G. (1993) Groundwater nitrate dynamics in grass and poplar vegetated riparian buffer strips during the winter. *Journal of Environmental Quality,* **22**, 273-278.

Howard-Williams, C., Davies, J. and Pickmere, S.E. (1982) The dynamic of growth, the effects of changing area and nitrate uptake by watercress *Nasturtium officinale* R. Br. in a New Zealand stream. *Journal of Applied Ecology,* **19**, 589-601.

Howard-Williams, C., Pickmere, S.E. and Davies, J. (1983) Decay rates and nitrogen dynamics of decomposing water cress *Nasturtium officinale* R. Br.. *Hydrobiologia,* **99**, 207-214.

Howard-Williams, C. and Downes, M.T. (1984) Nutrient removal by streambank vegetation. In: Wilcock, R.J. (Ed.) *Land Treatment of Wastes, Water and Soil.* Miscellaneous Publications. Government Printer, Wellington. pp 409-422.

Howard-Williams, C., Pickmere, S.E. and Davies, J. (1986) Nutrient retention and processing in New Zealand streams: the influence of riparian vegetation. NZ *Agricultural. Science,* **20**, 110-115.

Howard-Williams, C., Clayton, J.S., Coffey, B.T. and Johnstone, I.M. (1987) Macrophyte invasions. In: Viner, A.B. (Ed.) *Inland Waters of New Zealand.* DSIR Science Information Publishing Centre, Wellington, New Zealand. pp 307-331.

Howard-Williams, C. and Pickmere, S.E. (1994) Long-term vegetation and water quality changes associated with the restoration of a pasture stream. In: Collier, K.J. (Ed.) *Restoration of Aquatic Habitats.* Department of Conservation, Wellington, New Zealand. pp 171.

Lowrance, R., Velledis, G. and Hubbard, R.K. (1995) Denitrification in a restored riparian forest wetland. *Journal of Environmental Quality,* **24**, 808-815.

Mitsch, W.J. (1992) Landscape design and the role of created, restored and natural riparian wetlands in controlling nonpoint source pollution. *Ecological Engineering,* **1**, 27-47.

Myrold, D.D. and Tiedje, J.M. (1985) Diffusional constraints on denitrification in soil. *Soil Science Society of America Journal,* **49**, 651-657.

Osborne, L.L. and Kovacic, D.A. (1993) Riparian vegetated buffer strips in water-quality restoration and stream management. *Freshwater Biology,* **29**, 243-258.

Petersen Jr., R.C. (1992) The RCE: a riparian, channel and environmental inventory for small streams in the agricultural landscape. *Freshwater Biology,* **27**, 295-306.

Schipper, L.A., Cooper, A.B., Harfoot, C.G. and Dyck, W.J. (1993) Regulators of denitrification in an organic riparian soil. *Soil Biology and Biochemistry,* **25**, 925-933.

Sedell, J.R., Steedman, R.J., Regier, H.A. and Gregory, S.V. (1991) Restoration of human impacted land-water ecotones. In: Holland, M.M. (Ed.) *Ecotones: the role of landscape boundaries in the management and restoration of changing environments.* Chapman & Hall, London. pp. 110-129.

Smith, C.M. (1992) Riparian afforestation effects on water yields and water quality in pasture catchments. *Journal of Environmental Quality,* **21**, 237-245.

Tiedje, J.M., Sexstone, A.J., Myrold, D.D. and Robinson, J.A. (1982) Denitrification: ecological niches, competition and survival. *Antonie van Leeuwenhoek,* **48**, 569-583.

Tiedje, J.M., Simkins, S. and Groffman, P.M. (1989) Perspectives on measurement of denitrification in the field including recommended protocols for acetylene based methods. *Plant and Soil,* **115**, 261-284.

Triska, F.J., Duff, J.H. and Avanzino, R.J. (1993) Patterns of hydrological exchange and nutrient transformation in the hyporheic zone of a gravel-bottom stream: examining terrestrial-aquatic linkages. *Freshwater Biology,* **29**, 259-274.

Vance E.D., Brookes P.C. and Jenkinson, D.S. (1987) An extraction method for measuring soil microbial *C. Soil Biology and Biochemistry,* **19**, 703-707.

Walters, C.J. and Holling, C.S. (1990) Large-scale management experiments and learning by doing. *Ecology,* **71**(6), 2060-2068.

Warwick, J. and Hill, A.R. (1988) Nitrate depletion in the riparian zone of a small woodland stream. *Hydrobiologia,* **157**, 231-240.

Yoshinari, T. and Knowles, R. (1976) Acetylene inhibition of nitrous oxide reduction by denitrifying bacteria. *Biochemistry and Biophysical research Communications,* **69**, 705.

The Establishment of Buffer Zones – The Habitat Scheme Water Fringe Option, UK

25

A. Tytherleigh

ADAS Statutory Centre, Staplake Mount, Starcross, Devon, EX6 8PE, UK

Abstract

This paper describes the Habitat Scheme Water Fringe Option being piloted by the Ministry of Agriculture, Fisheries and Food in England. It outlines the background to the introduction of this experimental measure, describes how it is being operated in practice, reviews experience to date and indicates likely future developments.

In outlining the background it explains the role of the Habitat Scheme as part of the UK programme under the EC Agri-Environment Regulation (2078/92). The paper briefly describes the three components of the Habitat Scheme (Former Set Aside Land, Water Fringes and Saltmarsh), before concentrating on the Water Fringe Option.

The objectives of the Water Fringe Option are explained, followed by a description of the six pilot Water Fringe areas including how they were selected and their physical characteristics. The paper then describes how the Scheme operates in practice, including the buffer strip/extensive grazing options, payment levels, management prescriptions and how applications are processed. The arrangements for monitoring and evaluating the environmental impact of this measure are also outlined.

The paper then reviews progress to date, including uptake, and describes the outcome of the first two-yearly review of payment levels and other relevant matters. Changes in payment rates and other scheme rules introduced in April 1996 are described. The paper then concludes by outlining anticipated future developments with this pilot measure, including the proposals for a full evaluation and review in 1998/99.

INTRODUCTION

For many years the United Kingdom Government has been committed to conserving the country's best landscape, wildlife and historic features. This commitment, shared by most farmers in the United Kingdom, has been challenged in recent decades by the rapid pace of changes in agricultural technology and the economic pressures placed on farmers to adapt to these changes.

In response to concerns about the effects of agricultural intensification in the United Kingdom, the Government, with approval of the European Commission, set up a programme of environmental protection schemes in 1986. The success of the programme, the Environmentally Sensitive Areas (ESAs), partly inspired one of the measures accompanying the reform of the European Commission's Common Agricultural Policy in 1992 leading to the adoption of Council Regulation 2078/92, the "Agri-Enviroment Regulation". This regulation was a significant development because it made available, for the first time, Common Agricultural Policy funds to encourage environmentally sensitive agricultural practices. In the United Kingdom the regulation has given the Government an opportunity to expand the existing programme and to introduce a range of smaller pilot schemes which are seen as new and challenging approaches to environmental protection.

IMPLEMENTATION OF REGULATION 2078/92 IN ENGLAND

The Government's programme for the implementation of the agri-environment regulation in England was drawn up in consultation with other statutory environmental advisers. It was based on an

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0

assessment of the major environmental challenges and opportunities facing the English countryside. The Ministry of Agriculture, Fisheries and Food (MAFF) issued for public consultation in early 1993 proposals for a range of schemes (MAFF, 1993a). By spring 1994 the proposals had been finalised and the schemes had received formal approval from the European Commission. The schemes which make up England's agri-environment programme are listed in Table 1.

Table 1. Agri-Environment Schemes in England.

Environmentally Sensitive Areas
Countryside Stewardship
Nitrate Sensitive Areas
The Organic Aid Scheme
The Countryside Access Scheme
The Moorland Scheme
The Habitat Scheme

Six new ESAs were designated under the programme in early 1994. There are now 22 ESAs in England covering 10% of the agricultural area (MAFF, 1994a). The Countryside Stewardship Scheme (MAFF, 1996a), created in 1991, operates throughout England and aims to protect, enhance, restore and re-create targeted landscapes, their wildlife habitats and historic features and to improve opportunities for public enjoyment of the countryside. Twenty-two Nitrate Sensitive Areas were launched in July 1994 adding to ten existing areas (MAFF, 1994b). This scheme offers incentives to farmers to undertake significant changes in agricultural practices to reduce nitrate leaching in areas where groundwater sources are used to supply drinking water. The Organic Aid Scheme (MAFF, 1994c) is available to farmers throughout England who wish to convert to organic production. Launched in September 1994 the Countryside Access Scheme (MAFF, 1995a) aims to increase the benefits derived from land which is in set-aside by offering incentives to farmers to increase public access opportunities on the best located sites. The Moorland Scheme (MAFF, 1995b) aims to protect and improve the moorland environment outside of ESAs by encouraging upland farmers to reduce grazing where this will improve the condition of heather and other moorland vegetation. The Habitat Scheme (MAFF, 1996 b,c,d) was launched in May 1994 and offers incentives to farmers to manage their land to create or improve valuable wildlife habitats over 10 or 20 years.

THE HABITAT SCHEME

The Habitat Scheme is an experimental pilot scheme targeted at specific habitat types. There are three options in the Scheme, one of which, the Water Fringe Option, is described in detail in this paper.

The Saltmarsh Option *(MAFF, 1996b)*

This option aims to convert existing arable and permanent grassland situated adjacent to the English coast to saltmarsh. The main objective is to create important wildlife habitats capable of supporting a wide diversity of plants and animals and to take land out of agricultural production for at least 20 years. In the first two years of the scheme 60 hectares of land have been entered into this option.

The Former Set-aside Option *(MAFF, 1996c)*

From 1988 to 1991 MAFF offered farmers the opportunity to enter a Five Year Set-aside Scheme on arable land. Changes in the European Community's arable support arrangements led to this Scheme being closed in 1991. Monitoring revealed that some of the land in the Scheme had developed into important wildlife habitats which would be lost if the land was returned to agricultural production. The Former Set-aside Option enabled farmers to continue to protect and enhance these wildlife habitats, particularly in areas where they adjoined other important habitats. The agreements are for 20 years and there are currently 4200 hectares in this option.

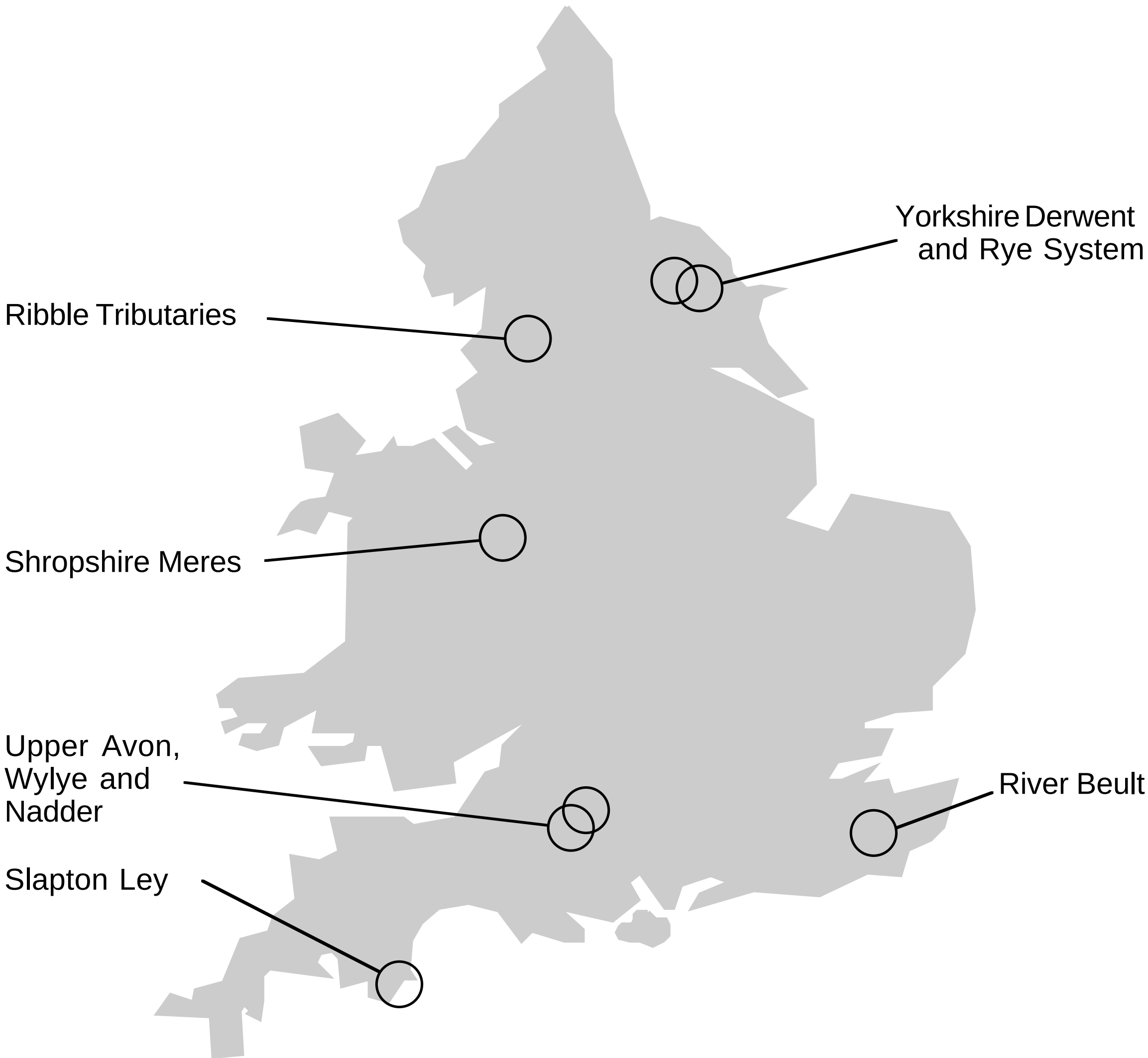
THE WATER FRINGE OPTION

Waterside habitats are considered to be important not only for supporting a wide range of plants and animals but also for the protection of fresh water from various forms of pollution. In England there has been general concern that the quality of waterside habitats is in decline (MAFF, 1993c). MAFF's overall objective for the Water Fringe Option (MAFF, 1996d) is to protect and enhance the wildlife value of watercourses and adjacent habitats by encouraging farmers to manage waterside land in an environmentally beneficial way. In addition the Water Fringe Option aims to contribute to the improvement of fresh water quality by reducing soil erosion and nutrient inputs from agricultural sources.

The Water Fringe pilot areas

The Water Fringe Option is targeted at six pilot areas (Fig. 1). These areas were chosen by MAFF following consultation with other statutory environmental advisers and after evaluation by ADAS (an Executive Agency of MAFF). The areas represent a cross-section of important English watercourses and waterside habitats. All are considered to be under threat or in decline mainly as a result of changes in agricultural practices within their catchments.

Figure 1. Water Fringe Pilot Areas



The *Derwent* pilot area in North Yorkshire includes part of the main river and its tributaries. The river system is a good example of a clay river with a transition to alluvial substrate in its lower reaches. Sections of the river are English Nature Sites of Special Scientific Interest (SSSI) with over 100 species of aquatic plants recorded together with a rich assemblage of invertebrates, fish species and a good population of otter. Many of the tributaries flow through intensive agricultural land. The increased pressures from intensive livestock grazing and arable cultivation have resulted in the loss of bankside vegetation and contributed to a rise in soil erosion, suspended solid inputs and increased nitrate loadings.

In Lancashire, the Swanside and Ings Becks are the tributaries included in the Ribble pilot area. The *Ribble* is a major salmonid river and both becks are valuable spawning areas for salmon and trout. In recent years improvement to adjoining grassland has resulted in an increase in livestock grazing, particularly by dairy cattle, along much of the length of the becks. There has been an increase in physical damage to the becks and as a consequence a decline in the fish populations.

The *Shropshire Meres* are nationally important open water sites, four of which have been selected to pilot the Water Fringe Option. These are Crosemere, Betton Pool, Fenemere and Berrington Pool. All are good examples of naturally eutrophic lakes with important aquatic and fringe habitats supporting a rich variety of invertebrates and birds. Designated SSSIs by English Nature, Fenemere and Berrington Pool also form part of the West Midlands Meres Ramsar Convention Wetlands of International Importance. Surveys in recent years of all the sites have shown an increase in concentrations of nutrients, especially nitrate, and a decline in sensitive aquatic plant species and fringe vegetation. There has been an expansion of arable cropping and more intensive livestock grazing in the mere catchments.

In Kent a 27 kilometre section of the River *Beult* forms the pilot area. The river is a good example of a clay river rising on sandstone. The flora along the whole river are diverse, including both submerged and emergent plants supporting a good range of invertebrates and birds. The river is also very important for course fishing. The character of the river and its corridor has been altered by man over the years but recent measures, including re-alignment, removal of bankside trees and installation of land drainage, have had a significant impact which, together with contributions from more intensive farming systems, have resulted in a marked reduction in water quality and a decline in the waterside diversity.

The *Upper Avon*, Wylye and Nadder rivers in Wiltshire, tributaries of the River Avon, form one of the best examples of a chalk river system in England. All three rivers have important communities of true aquatic plants providing a rich and varied habitat for numerous species of aquatic insects and fish. The damp species-rich meadows adjacent to the rivers are also important for breeding birds. Traditionally there was low intensity farming on the land adjacent to the rivers. In recent years improved land drainage and increased fertiliser use has enabled more intensive farming in the river corridor and there has been a loss of wetland and river margin habitats and a deterioration in river water quality.

The *Slapton Ley* pilot area in South Devon includes both the Ley and its associated catchment of 46 square kilometres. Designated a National Nature Reserve in 1993, Slapton Ley is the largest natural freshwater lake in South West England, supporting a rich diversity of flora and fauna. It is protected from the sea by a shingle bar and comprises the Higher Ley, 39 hectares of mainly rich fen and carr vegetation, and the Lower Ley, 77 hectares of shallow open water lake with reedswamp margins. The Ley is fed by a dense network of streams flowing into two main watercourses, the River Gara draining into the Higher Ley and the Start Stream into the Lower Ley. The agricultural land use in the catchment is dominated by mixed farming, the intensity of which has increased in recent decades. There have been increases in nutrient and sediment loadings and a corresponding decline in the water quality of the streams and of the Ley itself.

Implementation of the Water Fringe Option

The legislation currently governing the Water Fringe Option is The Habitat (Water Fringe) (Amendment) Regulation 1996 (MAFF, 1996e). As with most agri-environment schemes participation by farmers is entirely voluntary. Farmers with responsibility for agricultural land immediately adjacent to the designated watercourse may enter into 10- or 20-year agreements with MAFF. The Scheme is presented by MAFF as a long-term commitment to habitat improvement and, unlike some other agri-environment schemes, there is no opt-out from the agreements.

Farmers joining the Water Fringe Option are required to adopt management practices in addition to those which they would be expected to follow under the MAFF series of Codes of Good Agricultural Practice. These codes, for Air (MAFF, 1992), Soil (MAFF, 1993b), Water (MAFF, 1991) and the Safe Use of Pesticides (MAFF/HSC, 1990) are partially statutory and are designed to help to prevent damage and pollution.

Water Fringe management options

MAFF has produced an information pack for farmers which explains the Water Fringe Option (MAFF, 1996d). Farmers can apply to enter agricultural land into one of two management options. They may withdraw land from agricultural production, by taking strips of land out of production for 20 years and managing the strips to create wildlife habitats to act as buffers between neighbouring agricultural land and the water. Alternatively, they may manage waterside fields by extensive grazing for 10 years.

Withdrawal from agricultural production

Under this option the area taken out of production to create a buffer strip will have an average width of between 10 and 30 metres. However, in situations where fields adjacent to the designated water course are narrow, withdrawal of the whole field may be considered if that will further the purpose of the option. Up to 20 metres of non-agricultural land, for example wood or scrub, may also separate the buffer strip from the water's edge. Agreements entered into under this option are for 20 years.

To protect the buffer strip from encroachment by livestock the farmer is required to fence off the land withdrawn from agricultural production. Assistance in the form of a capital grant towards the cost of new permanent fencing is available under the Water Fringe Option. Where arable land is entered the farmer is asked to establish a permanent grass sward within seven months of joining the scheme. This may be done by sowing an agreed grass seed mix or by natural regeneration.

A management plan for the buffer strip is devised with the farmer. The plan will take into account the nature of the land, its location and its environmental potential. It will normally include one of three basic options :

(i) to allow the buffer strip to develop into scrub with no or minimal cutting

(ii) to mow all the vegetation on the buffer strip during August each year

(iii) a combination of (i) and (ii)

A management plan under options (i) and (ii) might include leaving a narrow strip of one to two metres uncut next to the watercourse or lake. Alternatively, it could involve cutting one half of the strip each year on a rotational basis. The plan drawn up at the start of the agreement will not be expected to remain fixed for 20 years. The aim is to design a flexible management scheme to take account of changes in the vegetation on the buffer strip and to achieve the greatest habitat diversity.

Extensive grassland management

Whole or part fields adjacent to the designated watercourse or lake may be entered into the extensive grassland management option. The land may be separated from the watercourse or lake by up to 20 metres of wood or scrub. Under this option the farmer must continue to graze the land with cattle, sheep or deer but must avoid overgrazing, undergrazing or poaching. To minimise the risk of damage to ground-nesting birds through trampling and to allow more wild flowers and grasses to set seed, the stocking rate during April and May must be kept within a range from 0.75 livestock units per hectare to 1.4 livestock units per hectare, and the land may be cut each year only after the end of June. If arable land is entered into this option a permanent grass sward must be established, either by sowing an approved grass species mix or through natural regeneration.

Agreements under this option are for 10 years. It is anticipated that by reducing the inputs to land under agreement the carrying capacity will gradually be lowered. The effect of this will be to encourage habitat diversity and to reduce the risk of soil erosion and nutrient loss into watercourses and lakes.

Standard management prescriptions

Under the Water Fringe Option there are a number of standard management prescriptions (Table 2) which the farmer is required to follow to complement the specific management prescriptions under the withdrawal and extensive grassland options.

Table 2. Standard Management Prescriptions

- No inorganic or organic fertiliser inputs
- No lime, slag or other substances used to reduce soil acidity
- No fungicide or insecticide applications
- Some control of non-indigenous, injurious weeds and bracken is permitted
- No storage or dumping of any materials or disposal of any pesticides on land under agreement
- Any tree or shrub planting must be agreed in advance
- Existing trees, shrubs and hedges on or bordering the land under agreement must be retained and managed
- Existing watercourses, ditches, ponds and reedbeds on or bordering the land under agreement must be maintained
- No installation of new land drainage systems or modification of existing land drains to bring about improved drainage
- All traditional buildings, stone walls or features of archaeological or historical value or interest must not be damaged, destroyed or removed
- Permission must be obtained before undertaking any building or engineering operations
- Additional public access on land under agreement must be agreed in advance with the Ministry

Raised water level supplement

In addition to the two main management options there is a supplement available to farmers to encourage high water levels on land entered into agreement. There are two methods by which farmers may achieve raised water levels. They may either intercept field drainage flows, to reduce the flow of waterborne nutrients and sediment into watercourse or lake, or alternatively maintain high water levels in ditches to create shallow pools.

Reducing the flow of nutrients and sediment into watercourses and lakes will benefit aquatic flora and fauna and help maintain the biological diversity of such waters. Maintaining high water levels in ditches from December through to April will benefit overwintering and breeding birds as well as aquatic fauna and flora. The creation of shallow pools will maximise the wildlife benefits by providing suitable conditions for feeding waders.

Raised water level supplements will last for either 10 or 20 years depending on the option the farmer has entered. However, before carrying out any work required the farmer must obtain consent from the appropriate water and drainage authorities. Account will also need to be taken of possible effects that raising water levels on agreement land may have on land outside the agreement area.

Payments

In return for agreeing to follow the management prescriptions farmers receive an annual payment according to the type of land entered into the scheme (Table 3). The payment rates are derived from calculations of income foregone and management costs involved in implementing the scheme. MAFF is committed to reviewing the rates every two years.

Table 3. Water Fringe Option Payment Rates £/ha

Withdrawal of permanent grassland from production	240
Withdrawal of arable land from production	485 (405)
Extensive grassland management on permanent grass	125
Extensive grassland management on arable land	435
Raised water level supplement	40

Payments are made to farmers only on land under agreement. Therefore under the withdrawal option payments on short and narrow buffer strips may be as little as £70 per year. Farmers entering arable land into the withdrawal from agricultural production option may count the area towards their set-aside commitment and receive a reduced payment of £405 per hectare. Payments under the extensive grassland management option will be on whole or part fields entered. The raised water level supplement is an additional payment to any of the others.

Management of the Water Fringe Option

Each pilot area is managed by a qualified ADAS Project Officer who is generally the farmers' first point of contact. The Project Officer plays an important role in promoting the Water Fringe Option and encouraging and persuading farmers to adopt the long-term management requirements. For MAFF the Project Officer assists in the processing of applications and evaluation of the scheme. Once an agreement is set up the Project Officer provides a valuable link between the farmer and MAFF by providing advice and guidance on the practical aspects of implementing the scheme. Administration, enforcement and payment is carried out by MAFF's regional organisation, as for other agri-environment schemes.

Uptake of the Water Fringe Option

In the first two years of the pilot Water Fringe Option 112 farm agreements have been established. Uptake of the scheme has been variable (Table 4).

Table 4. Length (Km) of Bankside in agreement after 2 years.

Pilot Area	*Buffer Strips*	*Extensive Grassland*	*Total*	*% Eligible Bank*
Derwent	6.80	21.24	28.04	9.5
Beult	0.90	7.52	8.42	16.2
Ribble	0.69	5.39	6.08	12.2
Meres	0	0	0	0
Avon	5.45	40.53	45.98	19.4
Slapton	1.14	8.66	9.80	6.0
Total	14.98	83.34	98.32	12.3

There are now nearly 100 kilometres of bankside under the Water Fringe agreement, accounting for just over 12% of the eligible length. MAFF has set a target of 30% of eligible bankside to be entered by the end of the experimental pilot phase in 1998/99. Several of the pilot areas are well on the way to achieving this target, in particular the Upper Avon area. In the Shropshire Meres pilot area the small number of farmers with eligible land have not yet been persuaded to join the scheme. Within the Slapton Catchment pilot area 2 kilometres of Slapton Ley's eligible unbuffered margin is now under agreement.

Overall the extensive grassland management option has been more attractive to farmers. All the bankside taken up in this option would have included whole fields. Uptake of the withdrawal option has been more limited. In some of the pilot areas, for example Slapton and the Ribble Tributaries, the buffer strips option has proved less popular mainly because it restricts direct access by stock to the water, there being no capital payment available under the Scheme to provide alternative drinking water supplies.

The monitoring programme

A programme of monitoring is being carried out by ADAS to provide information on the environmental impact of the Water Fringe Option and to assess whether the main objectives of the scheme are being achieved. As the Scheme is principally concerned with the creation and enhancement of habitats, the monitoring objectives and evaluation criteria have been focused on ecological aspects.

There are three themes to the monitoring programme, two botanical and one physical. The first botanical assessment is of the composition of the buffer strip or whole field under extensive grassland management. Data on the range and frequency of grass and herb species is recorded in 30 objectively-positioned half-metre quadrats.

The second botanical assessment is of the bankside vegetation. The “bank” is defined as the area which slopes from the mean field level down to the water’s edge. The assessment is on a representative 20-metre section of the bank which is permanently marked by underground metal markers and measurements to fixed points. The section is divided into five 4-metre sub-sections and vegetation rooted on the bank is recorded to species level.

The final monitoring is an assessment of the impact of muddy access points created by livestock on the banks of the watercourse. A “muddy access point” is defined as an area of bank on which hoof prints can be seen in the mud. Such areas contribute to silt deposition which can have an impact on water quality. The monitoring is carried out by recording the presence or absence of muddy access points wherever livestock have access to the water frontage. In some situations this may be unrestricted over the whole length of the field bank; in others, fencing may limit access to one or more clearly defined points. Where stock have unrestricted access the length of water frontage is measured on a 1:2500 plan and the distance divided into 30 equal sections. The bank is walked and the presence or absence of a muddy access point is recorded for each section. This is used to give an index of erosion caused by livestock. If access is controlled by fencing, the location of the access points are marked on a site map and the length of bank over which the stock have access measured.

The monitoring programme to establish baseline data on the botanical composition of the buffer strips and fields was started in 1995 and has been completed this year. The sites will be re-surveyed during 1997 and 1998.

REVIEW OF THE HABITAT SCHEME

In autumn 1995 MAFF undertook the first review of the payment rates and other aspects of the Habitat Scheme in the light of uptake levels and experience gained in the first two years of operating the scheme. Within the Water Fringe Option MAFF considered that there had been a good level of farmer uptake of the extensive grassland management option, particularly on permanent grass. However, uptake of the withdrawal option and the arable land intake had been lower than originally anticipated. Following advice from ADAS and other statutory environmental organisations MAFF considered that changes to the payment rates and some management prescriptions and other rules were necessary to increase the effectiveness of the Water Fringe Option in fulfilling its environmental objectives.

Evaluation, by ADAS, of arable incomes showed that the original payment rates for the arable options under the Water Fringe Option did not adequately account for the income foregone by farmers with either buffer strips or land under extensive management. To rectify this MAFF increased the payment rates under the arable options from £360 and £260 per hectare to £485/405 and £435 per hectare. MAFF also negotiated with the European Commission the option for arable land entered into the withdrawal option to count towards farmers' set-aside commitment.

When the Water Fringe Option was launched in 1994 the capital grant available to farmers for fencing was administered under a separate MAFF Farm and Conservation Grant Scheme. Following the review and advice from ADAS on the costs of appropriate stockproof fencing the grant has now been integrated into the Water Fringe Option.

Following feedback from the ADAS Project Officers, a number of changes were made to the management prescriptions. These included a more flexible approach to the spring stocking rate and the addition of deer to the list of species permitted to graze land under extensive management. Discretion to allow farmers entering the arable options to retain existing ley grassland rather than require reseeding was introduced. In several of the pilot areas, particularly Slapton Ley, there was concern that the requirement to plough up ley grassland would increase the risk of nutrient loss and soil erosion into adjoining watercourses. It was recognised that this was in direct conflict with the ecological and water quality objectives of the scheme. The last change to the scheme introduced by MAFF was the option to allow farmers to enter part of a field into the extensive grassland management option.

DISCUSSION

The Habitat Scheme, introduced under the Agri-Environment Regulation 2078/92, has been running in England for two years. The Scheme is one of a number of new and challenging initiatives set up by the United Kingdom's Ministry of Agriculture Fisheries and Food to address the environmental concerns about the effects of agricultural intensification in vulnerable areas of high landscape and wildlife value.

The Habitat Scheme introduced three options, one of which, the Water Fringe Option, has been reviewed in this paper. The Water Fringe Option is an experimental pilot scheme operating in six pilot areas in England. The Option is one of the first national schemes of its kind aimed at establishing buffer zones on agricultural land. MAFF's main objective of the Water Fringe Option is habitat creation and enhancement and it is seeking to achieve this by formulating management agreements with farmers for 10 or 20 years.

In the two years the Water Fringe Option has been operating 112 agreements have been set up covering 800 hectares of agricultural land and 12% of the eligible bankside. There are two management options available under the scheme and of the two the 10-year extensive grassland management option has proved more attractive to farmers rather than the 20-year buffer strips option. The reasons for this can be related to both the lengths of the two agreements and to the management prescriptions. Farmers appear to be reluctant to commit themselves to long-term agreements, particularly where there is no opt-out clause. The watercourses are, in some of the pilot areas, an important feature of the farm, providing a cheap water supply for stock. Establishment of a buffer strip would effectively cut off the supply and incur additional costs in providing alternative drinking water, a cost which many farmers are unable or unwilling to bear.

In response to levels of farmer uptake and to feedback from ADAS and other organisations MAFF has already introduced changes to the Habitat Scheme. It is envisaged that these changes will attract more farmers into the scheme. In the case of the Water Fringe Option MAFF has increased substantially the payment rates for arable land brought into agreement and has taken a more flexible approach to spring stocking rates and other management requirements.

As with all agri-environment schemes MAFF is committed to carrying out a full evaluation of the Habitat Scheme Water Fringe Option at the end of the pilot phase in 1998/99. The evaluation will be based on the level of farmer uptake and on the results of the environmental monitoring programme currently being undertaken. At present it is too early in the life of the Water Fringe Option to predict the future of the Scheme but it is possible that it may be integrated at the end of the pilot phase with another main environmental scheme, the Countryside Stewardship Scheme.

ACKNOWLEDGEMENTS

Advice and information received from L. Baker and M. Barbier of MAFF Conservation and Woodland Policy Division and from R. Symes, S. McKenzie and the ADAS Water Fringe Project Officers is gratefully acknowledged.

REFERENCES

HSC/MAFF (1990) *Pesticides: Code of Practice for the Safe Use of Pesticides on Farms and Holdings.* HMSO, London

MAFF (1991) *Code of Good Agricultural Practice for the Protection of Water* MAFF Publications, PB 0587, London.

MAFF (1992) *Code of Good Agricultural Practice for the Protection of Air.* MAFF Publications, PB 0618, London.

MAFF (1993a) *Agriculture & England's Environment, A Consultation Document on Environmental Schemes under the Common Agricultural Policy (CAP),* MAFF Publications, London.

MAFF (1993b) *Code of Good Agricultural Practice for the Protection of Soil.* MAFF Publications, PB 0617, London.

MAFF (1993c) *Habitat Improvement Scheme, A Consultation Document in Agriculture & England's Environment.* MAFF Publications, London.

MAFF (1994a) *Environmentally Sensitive Areas Information Pack.* Revised Edition. MAFF Publications, PB 0943, London.

MAFF (1994b) *The Nitrate Sensitive Areas Scheme Information Pack.* MAFF Publications, PB 1729A, London.

MAFF (1994c) *The Organic Aid Scheme Information Pack.* MAFF Publications, PB 1825/1, London.

MAFF (1995a) *The Countryside Access Scheme Information Pack.* Revised Edition. MAFF Publications, PB 1896A, London.

MAFF (1995b) *The Moorland Scheme Information Pack.* MAFF Publications, PB 2058, London.

MAFF (1996a) *The Countryside Stewardship Scheme Information Pack.* MAFF Publications, PB 2505, London.

MAFF (1996b) *The Habitat Scheme Saltmarsh Information Pack.* Revised Edition. MAFF Publications, PB 1731A, London.

MAFF (1996c) *The Habitat Scheme Former Set-Aside Land Information Pack.* Revised Edition. MAFF Publications, PB 1732A, London.

MAFF (1996d) *The Habitat Scheme Water Fringe Areas Information Pack.* Revised Edition. MAFF Publications, PB 1730A, London.

MAFF (1996e) *Statutory Instrument No. 1480 The Habitat (WaterFringe) (Amendment) Regulations 1996.* HMSO, London.

The potential impact of buffer zones in agricultural practice

26

R.J. Cook

Morley Research Centre, Morley, Wymondham, Norfolk NR18 9DB, UK

Abstract

The Morley Research Centre farm is a typical East Anglian arable unit on a sandy loam (Ashley series) soil over chalky boulder clay. The farm is 50 m above sea level and straddles the watershed between the North Sea and the Wash to the west. It covers 200 of the 370 ha farmed by the Centre, 180 of which are in arable production. The farm comprises 26 arable fields, 22 of which have open drainage ditches on at least one boundary. Inclusion of a 6 m buffer reduces the effective area for crop production by 6.3 ha (3.5%) of the effective cropping area. The area represents over 10% of the effective cropping area of three fields. The impact of the adoption of buffer zones assuming current crop production techniques and restriction of products from the 6 m headland is estimated to be worth 1.0% of the cereal output. Adoption of buffer zones for all crops costs £2000 or 1.3% of our current gross output. This figure represents about 6% of the 1994 net farm income for arable, mainly cereal farms in Eastern England.

There are additional unquantifiable costs involved in compliance with the buffer zone restrictions on pesticides. Some fields on this farm are kept small because it is convenient for some of our experimental work. We have also adopted an extensive hedge planting programme during the last five years to enhance the farm environment and its wildlife habitats. If this was a commercial operation we would remove one of our three recently planted hedges, and four established hedges to amalgamate fields into larger blocks and mitigate the economic impact of buffer zones. This operation would also entail piping three small ditches. In practice this would reduce the wildlife benefit of the farm and increase the adverse impact on the landscape. The very opposite of the effect that much environmental legislation is intended to have.

INTRODUCTION

Modern arable production has evolved rapidly to adopt new technologies and incorporate advances in the understanding of crop growth into a simple effective management system. In the last 50 years, for example, cereal production in Britain has been transformed. It has moved from a basic rotation controlled system with alternating non cereal crops giving average yields of 1.8 t/ha to an intensive, but still often rotationally cropped, and efficient system with national average wheat yields of over 7.5 t/ha. The top 10% of farms in eastern England produce average yields of 8.86 t/ha (Murphy, 1996) and on many farms, including Morley yields may sometimes exceed 10 t/ha (unpublished).

This change has had considerable impact on our countryside. The rapid expansion in production has been almost directly responsible for an increasing awareness by the public at large of the adverse effects the industry has imposed on our landscape. Many of these are more perceived than real, but responsible farmers are aware of these concerns. Positive steps are now made by the industry to demonstrate that modern agriculture can be environmentally benign.

The public concerns are firmly on the political agenda. They have led to many schemes which seek to soften the impact of agriculture and focus away from production to the maintenance or restoration of environmental features (e.g., Hedgerow Incentive Scheme, local hedgerow planting grants). The LEAF initiative (Linking the Environment And Farming) has developed from these concerns and aims to demonstrate not only that modern farming can take care of and improve the environment, but that Integrated Crop management (ICM) is a viable, modern and sustainable system. Morley is pleased to be associated with LEAF as one of the network of Demonstration Farms which show ICM in action. We are also pleased to be associated with FWAG (Farming and Wildlife Advisory Group) who can provide advice on buffer strip management.

Other changes seek to reduce perceived pollutants such as fertilisers or pesticides, excessive or irresponsible use of which will have adverse effects, like any input to any industry. The problem is, however, more acute in the case of agriculture, because all inputs are used over such a large land area.

Buffer zones are one attempt to reduce risks of water contamination and subsequent adverse effects on both human consumption and environmental pollution. They have arisen by application of the precautionary principal on the basis that potential damage will be limited if use is restricted in sensitive areas. For this reason there is little evidence as to how wide they should be or of what they should comprise. The Environment Agency has published a guide to buffer strips and how they may be used (Anon, 1996a). This paper considers the impact of their implementation on modern arable systems.

RESTRICTED PRODUCTS

The primary purpose of the buffer zone in agriculture is usually the protection of water quality and to reduce the risk of impact on humans. For this reason there are restrictions of use of nitrogen, for example near water courses (Anon, 1991). They do also protect sensitive fauna and can indirectly reduce the impact of pesticides and fertilisers on biodiversity.

Forty active ingredients have restrictions on use within 6 m of any water course and the products listed in the UK Pesticide Guide (Anon, 1996b) are shown in Table 1. Of these products two are fungicides specific to potatoes. Most of the others are likely to be used primarily on wheat, barley or oilseed crops, although they do have other uses. The pyrethroid insecticides are likely to be used to control the aphid vectors of barley yellow dwarf virus in cereals. The herbicides, except atrazine and simazine, could be used in cereal production, although their uses are unlikely to be widespread or general.

The logic and reasons for prohibition does need to be rationalised. For example, four of the listed insecticides belong to the relatively low toxicity group of pyrethroids whereas chlorpyrifos is highly toxic and in cereals is only likely to be used during the summer in the rare event of attack by one particular insect pest, the wheat orange blossom midge *(Sitodiplosis mosellana).* There was a severe outbreak in 1993 (Oakley, 1994) although evidence, a literature review by Oakley, suggests that severe development of the pest occurs about once every 20 years.

For most of these products there are alternatives, although in many circumstances their use could result in less effective control of the target organism and/or increased cost.

CROPPED BOUNDARIES

Most modern farms employ spray machines with 12, 18, 20 or 24 metre booms. It is possible to isolate part of these booms, to avoid spray treatment of the area beneath that part of the boom. Switching off the 6 m band alongside a stream or field margin is feasible, but does necessitate a return visit to the unsprayed area in order to apply any acceptable alternative product, if an attempt is to be made to control the target problem. In addition to the inconvenience of going back over the same area with another treatment there is the added complication of product choice and the associated reduced work rate – which also risks missing any suitable weather window for spraying.

Winter cereals

Barley yellow dwarf virus is transmitted by aphids and may be effectively controlled by insecticides proscribed from use in the 6 m band nearest a water course (Table 1). The disease is associated with sowing before mid-October, so there is a substantial risk of infection over 60-70% of the UK cereal crop which is sown during late September or early October. In the absence of an effective warning scheme most crops are routinely treated usually during early November. Current control strategies are a result of extensive work over many years although there is still considerable effort being invested in risk forecasting (e.g. Mann *et al.*, 1995). Even if an effective infection risk forecast scheme existed it is likely that losses would be considerable, simply because the only effective insecticide cannot be used on field margins where the risk that aphid colonies will establish is greatest. There is, however, much work in progress to improve the quality of advice to farmers and provide the necessary decision support to minimise crop treatments. There may be an additional option for control of this disease. Cereal seed treated with imidacloprid (Gaucho Bayer) would reduce autumn spread of virus but would not obviate the need for other treatments later in the season.

Table 1. Active ingredients for which use is restricted within 6 m of a water course

Fungicides	*Insecticides*	*Herbicides*
Anilazine	Alpha-cypermethrin	Atrazine
Chlorothalonil	Chlorpyrifos	Benazolin+bromoxonil+ioxynil
Dimethomorph+mancozeb	Cypermethrin	Bromoxonil + ioxynil
Epoxiconazole	Deltamethrin	Bromoxonil+ioxynil+mecoprop P
Epoxiconazole+fenpropimorph	Dimethoate	Bromoxynil+ioxynil+triasulfuron
Epoxiconazole+tridemorph	Lambda-chyalothrin	Dicamba-triasulfuron
Fenbuconazole+fenpropimorph	Deltamethrin+heptenophos	Fluoroglycofen-ethyl+mecoprop P
Fenpropidin+prochloraz	Esfenvalerate	Fluoroglycofen-ethyl+triasulfuron
Fenpropidin+tebuconazole	Fenpropathrin	Linuron
Propiconazole+tebuconazole	Quinalphos	Linuron+trifluralin
Fluazinam	Tebufenpyrad	Mecoprop P + triasulfuron
		Simazine
		Simazine+trietazine
		Chlorpropham+linuron
		2, 4-DB+linuron+MCPA
		Metsulfuron-methyl
		Metsulfuron-methyl+ thifensulfuron-methyl
		Thifensulfuron-methyl
		Thifensulfuron-methyl+ tribenuron- methyl
		Triasulfuron

Delaying of sowing or acceptance of the loss around each field, with the subsequent risk to the remainder of the crop as the infestation moves into the field, are the only viable cropping alternatives. Delay of sowing appears to be feasible, but the 6 m strip adjacent to any water courses must then be cultivated and sown after the adjacent crop has established, leading to damage and loss of potential in an area adjacent to the buffer strip. The option is untenable on most farms simply because of time constraints and the risk that crop damage or failure in the buffer strip will occur.

Winter wheat yields are reduced as sowing date is delayed from mid-September, which is generally considered optimum, until mid-December (Fielder, 1988). Data generated at Morley (Nuttall and Stevens, 1985) confirm the reduction in yield associated with delayed sowing date and illustrate the need to apply autumn insecticides for control of barley yellow dwarf virus and fungicides on early sown crops (Table 2). Subsequent work at Morley (unpublished data) has indicated that autumn fungicides are not justified on wheat but can be of benefit on winter barley (Stevens and Yarham, 1981). The increase in severity of some wheat diseases in early sown crops can be identified in surveys (Polley and Thomas, 1991; Table 3).

Table 2. Effect of sowing date on yield of second wheat at Morley, 1982-1984 (t/ha)

	Sowing Date			
Treatment regime	*Mid Sept.*	*End Sept.*	*Mid Oct.*	*Late Oct.*
Full autumn protection[1]	9.88	9.73	9.41	8.43
Insecticide*	9.88	9.51	9.13	8.25
Fungicide**	9.71	9.62	9.29	8.29
Late spring and summer only[2]	9.43	9.47	9.06	8.34
Sed		0.102		

[1] Insecticide and fungicide
[2] Full pest and disease control regime in spring and summer; spring and summer treatment was also applied to the other treatment regimes.

* Decis (deltamethrin, 25 g/l; 0.3 l/ha)
** Tilt (propiconazole, 125 g/l; 0.5 l/ha)

Table 3. Effect of sowing date on eyespot and take all in second wheat crops (MAFF surveys, 1982-1989)

	Mid Sept	*Late Sept*	*Mid Oct*	*Late Oct*	*Sed*
Take all (% plants GS 31)	9	33	10	14	-
Eyespot*	23.7	9.8	6.7	1.4	2.08
Sharp eyespot*	28.8	18.7	11.5	3.9	3.5

* % tillers affected in early July.
Data from Polley and Thomas, 1991

The relevance of this to buffer zones relates to choice of actions. Earlier sowing has the positive advantage of increased yield but, because of the increased disease risk, increases the likelihood of severe disease. Early sowing does have the benefit that the growing crop absorbs nitrogen during the autumn, thus reducing the amount of mineral nitrogen in the soil, decreasing leaching into ground water or entry to water courses through field drains (Powlson and Davies, 1993). Date of cultivation is the critical factor affecting nitrate mineralisation and a delay until late autumn is an effective method of nitrate management (Stokes *et al.,* 1992), but is not acceptable on heavier soils if a winter crop is to be established. That might restrict the buffer zone to spring cropping, an option which is also likely to be untenable on most arable farms.

Early sowing also increases the potential for severe weed infestations which at present may be controlled effectively by herbicides although there are now cases of resistance to commonly used products. Black-grass *(Alopecurus myosuroides)* and barren brome *(Anisantha sterilis)* can both cause severe weed problems in winter cereals, the latter often especially on field margins. The problem is exacerbated in the case of barren brome which is not well controlled by herbicides (e.g. Went, 1990). Although late sowing can reduce black-grass and is a management option where resistance occurs these interactions complicate further buffer zone management.

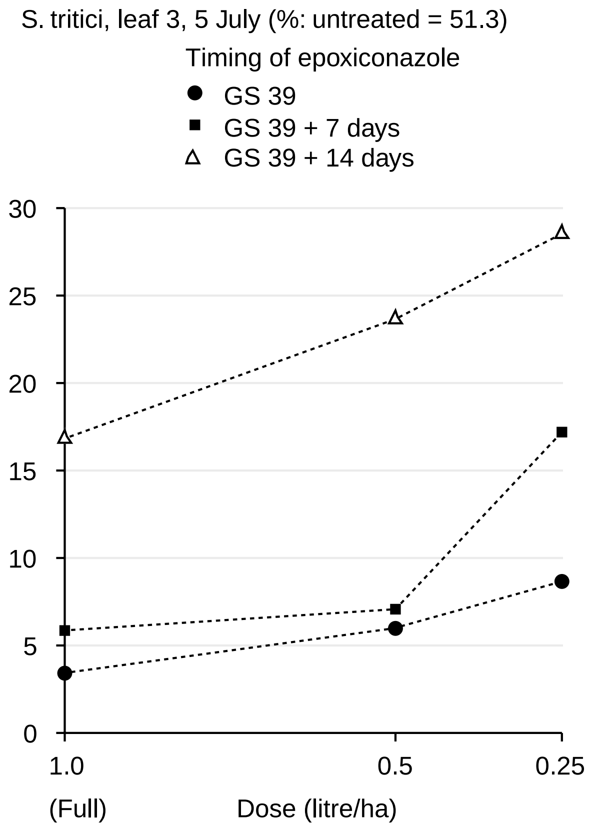
Application opportunities
There are specific guidelines for application of pesticides and on modern, large farms the weather often limits the number of spray opportunities. In order to combat these difficulties considerable work is now undertaken to optimise spray applications in relation to spray quality and pesticide timing.

Timing

Recent work at Morley (unpublished data) has indicated that relatively low doses of some products provide effective control of the septoria leaf blotch disease of wheat *(Septoria tritici)* if they can be applied within a few days of infection. As treatment is delayed the dose required to achieve comparable disease control increases (Fig. 1).

This can have considerable implications for the buffer zone and crop economics, particularly if adverse weather prevents buffer zone treatment for more than a few days. It is, incidentally, ironic that an effective treatment using a very low dose of some products can not be applied when a full dose of less effective but alternative products is allowed within the buffer zone.

Figure 1. Effect of application date and dose on control of *Septoria tritici* by epoxiconazole



Spray quality

Most pesticides are now applied as medium volume (*circa* 200 l water/ha) using a medium spray quality. It is well established that drift of spray can be influenced by droplet size and pressure. In some situations, for example weed control in sugar beet, a fine quality spray will improve activity (May and Hilton, 1992). These fine sprays can drift and may thus affect field margins. The interactions between spray quality, pesticide activity and environmental side effects need to be considered in all ICM systems. They have a potential impact on the buffer zone. However, there is a range of measures which can be taken to reduce drift, including additives, air assistance and increased droplet size. These may be appropriate when they do not adversely affect biological activity and may need to be considered in relation to product choice and timing.

Conservation headlands

ICM depends upon full exploitation of all the information needed in any crop management decision and consideration of the environmental impact of those decisions. Delayed sowing of headlands adjacent to water courses is, therefore, at least ostensibly, an attractive option simply because it reduces the need for some but not all proscribed pesticides. It would not remove the need for nitrogen or other fertilisers, except as a voluntary act by the individual farmer. Where the buffer zone is cropped the requirement to produce the best crop possible remains. Recommendations for the management of field margins, incorporating the boundary, a boundary strip and a part of the crop headland have been developed by the Game Conservancy Trust. Their aim is to encourage broad-leaved weed species and insects which provide food for game birds and other farmland birds. The recommendations (Anon, 1995) suggest that the area of crop nearest to each field boundary should be cropped but untreated with insecticides after March 15 and treated with only selected herbicides during the season. Application of this principle to the boundary strip adjacent to water courses or ditches is feasible. It does allow some crop production but will inevitably lead to increasing weed infestations and the risk of severe disease or pest attack. Justification is simply in terms of wildlife benefit. In addition to the risk that invasive weeds will become established there is the additional risk of yield loss from untreated annual weeds. Weed growth in cereals and sugar beet is negatively correlated with early ground cover by the crop (Lotz *et al.,* 1995). Although these results confirm the need to maximise early crop growth in order to minimise losses, early growth of invasive or competitive weeds will compete with any cereal crop. Weed competition argues against the use of untreated strips around field margins.

Omission of nitrogen applications on the boundary strip would have a greater impact. Cereal yields would be expected to fall by about 50% in the first year without nitrogen (e.g. Stevens and Nuttall, 1993) and further to about 2 t/ha within two or three years, imposing a considerable economic cost. In addition, there is likely to be a significant adverse effect on grain quality, as measured by protein, Hagberg falling number, SDS sedimentation value and specific weight (Nuttall and Stevens, 1991, Smith *et al.,* 1990) compounding the economic impact of compliance. These effects would spread beyond the no treatment zone if it were to be rigidly applied, imposing some further cost. Farms with pneumatic fertiliser spreaders could restrict applications with relative ease; spinning disc applications assume there will be overlap between adjacent treated bouts. In this case there may be a treatment gradient on the buffer zone, although a cut-off plate is normally used to restrict unnecessary treatments.

UNCROPPED BOUNDARIES

The additional complexity of crop management within the buffer zone raises options for leaving boundary strips alongside water courses uncropped. The opportunities for this depend upon circumstances. It is unfortunate that current set aside regulations only allow a 20 m strip of uncropped field margin. In the case of large fields, or where environmentally a wide margin is desirable, set-aside does represent an attractive option. It also allows the farmer to “straighten out” the curved or meandering margins of rivers. This can facilitate cropping by allowing standard farming practice right up to the boundary strips, minimising headland losses. It provides excellent opportunities for wildlife, especially if any necessary trimming can be completed in the autumn or winter.

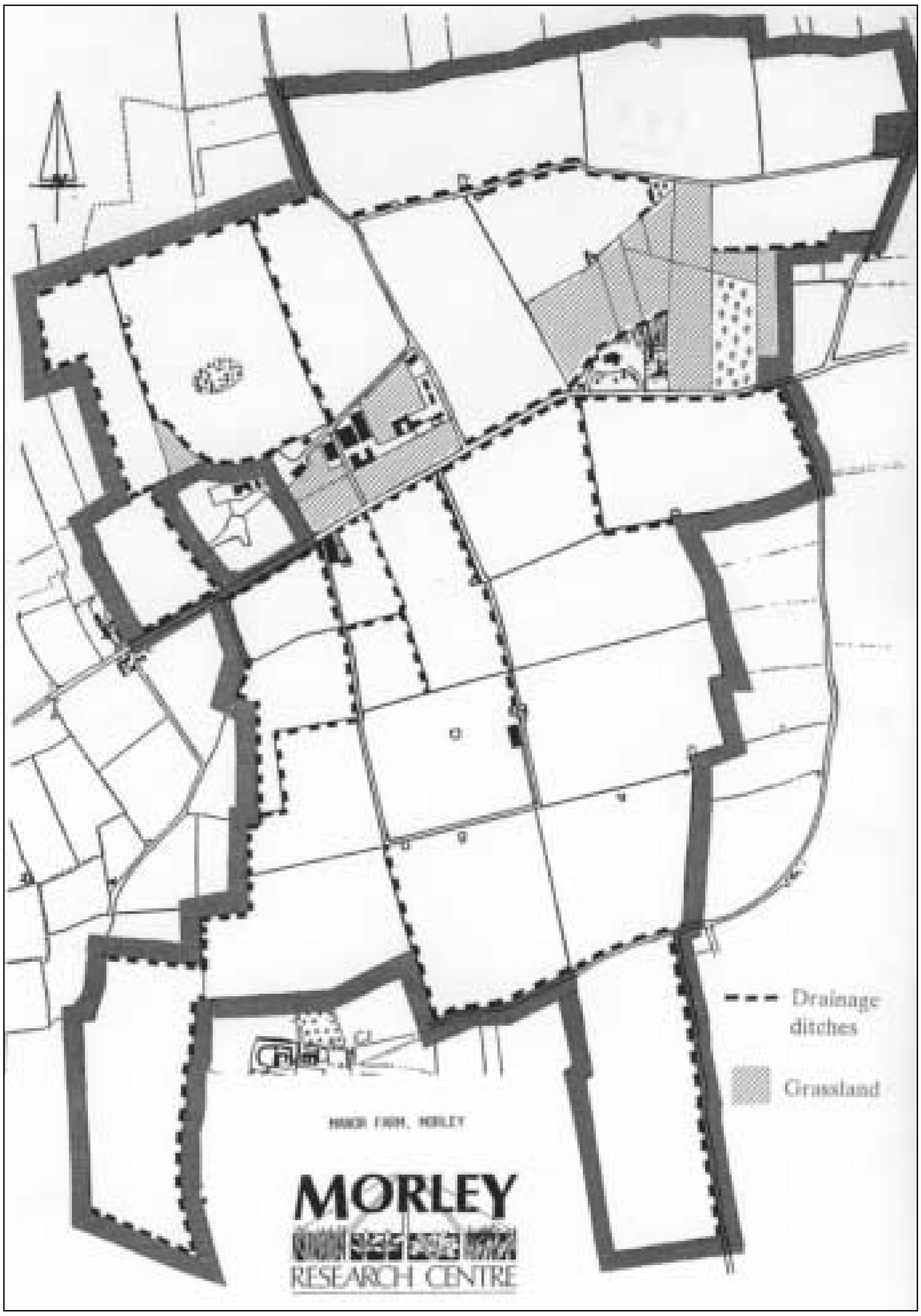
Treatments of buffer strips require an additional management decision to minimise possible environmental damage to the crop edge and restrict possible treatments in that crop area. ICM also requires consideration of the environment and that treatments are applied according to need, and on the basis of justifiable risk assessments. For this reason good pest management often, ironically, identifies field margins as the area where treatment is most likely to be necessary. In these cases the crop area closest to the hedge, grassland or water course is the area of highest pest risk, for example, for barley yellow dwarf virus vectors (mainly *Rhopalosiphum padi* or *Sitobion avenae*), grain aphids on wheat, pollen beetle (e.g. *Meligethes aeneus*) on oilseed rape. Good practice dictates that the headland alone is often the only part of the field to be treated. Many of the products we need to use in this boundary zone are proscribed within the 6 m buffer zone for a water course. For this reason we still need an untreated buffer and this could be uncropped.

The Countryside Stewardship Scheme for 1996 (Anon, 1996c) included an option allowing uncropped 6 m wide boundary strips. This was a potentially very attractive, funded, option for management of the buffer zone. The scheme was only available if the first 6 m of adjacent crop was managed in accordance with Game Conservancy headland guidelines.

CASE STUDY, MORLEY MANOR FARM

It is, perhaps, invidious to select any individual farm for particular scrutiny. However, Morley Manor Farm, Wymondham (Morley) is a demonstration farm. It is the home farm of Morley Research Centre and is located about 20 km south west of Norwich at about 50 m above sea level. The soils are sandy loams (Ashley series) which with the similar sandy clay Beccles soils represents about 45% of East Anglia. The farm is on the watershed between the North Sea and the Wash. Our field experiments are located within commercial crops.

Figure 2. Map of Morley Manor Farm showing drainage ditches requiring buffer zones



It is a typical East Anglian arable farm and can be taken to represent the impact of buffer zones. A plan of the farm is given as Fig. 2. The 200 ha farm comprises 26 fields, 22 of which have open drainage ditches on at least one boundary. There are about 14 ha of grass. The total length of ditches is about 9.0 km which, including water courses separating fields, represents 6.3 ha or 3.5% of the total cropping area. The restricted list of chemicals which can be applied to this area have been listed in Table 1. By using alternative products where available, we have minimised the effects on production but estimate that buffer zones reduce farm income by an average of £2000 per year, or 1.3% of gross output. If Morley is taken as an average farm this loss represents 6% of the average net farm income (£178/ha) for mainly cereal farms of 300-400 ha size in eastern England (Murphy, 1996). Some of these impacts are summarised in Table 4.

Table 4. Estimated impact of buffer zones on output for Morley Manor Farm

Crop	*Area of crop (ha)*	*Area of buffer zone (ha)*	*Estimated crop loss in zone (%)*	*Value of yield loss (£)*
Cereals	105	3.6	25	727
Sugar beet	40	1.4	50	1330
Linseeds and beans	40	1.4	25	111

Note: Assumes adverse effects are not cumulative in successive years.

The Morley farm includes several relatively small fields, which are convenient to retain because of our experimental work and for their value in maintaining the landscape. Continued restriction of buffer zones does, however, prompt consideration of the possible amalgamation of fields and the removal of hedges, simply to retain the benefit of more efficient operations. This would have the opposite effect to that desired. Modifications to field shapes would also allow more efficient field operations. In combination with grassed or similar uncropped field margins around ditches and water courses, it would simplify operations and reduce headland wastage. The landscape would be changed so that, unlike the original structure where crops abut onto hedges, there would be grass margins to fields which would produce wildlife refuges, but allow conventional farming within the remaining area of each field. Such grass areas, if mown once or twice during the year in September or in early spring, might provide improved habitat for many birds including barn owls and create wildlife corridors.

DISCUSSION

This paper has illustrated some of the difficulties associated with management of buffer zones. A summary of some advantages and disadvantages is given in Table 5. Whilst the effects are agriculturally negative, they do provide habitat management opportunities. It is inevitable that a production based industry will emphasise the disadvantages of any restrictions. Buffer zones do present considerable potential economic penalties, especially in areas with small fields and numerous ditches. In other areas the opportunities provided to access fields, create wider and more effective wildlife corridors and grassland habitat are likely to be of considerable environmental benefit. They have the potential to allow the application of a sound ICM system to extensive areas of arable production if they are managed sympathetically. It is inevitable that restrictions on any agricultural inputs will impact on crop production. Restricting nitrogen will reduce outputs directly and immediately, although full effects will become apparent over subsequent seasons.

Table 5. Advantages and disadvantages of buffer zones

Environmental benefits
- Grass strips maintain or increase biodiversity of field margin if subject to annual mowing and are of mixed species
- Restrict movement of toxic products into surrounding habitats
- Protect water courses from pesticides and nitrogen – the primary aim
- Grass or uncropped strips facilitate access for winter hedging operations without crop loss
- Restrict arable production to centres of fields away from habitat
- Facilitates control of invasive weeds, especially if 1 m bare soil strip maintained between buffer and crop

Agricultural disadvantages
- Loss of production and revenue
- Increased loss of efficiency in small fields
- Restrict land use
- Cropped buffer zones increase complexity of planning and execution of production and lose timeliness for farm operations
- Advance potential front of invasive weeds into field

Buffer strips are now part of arable production but they do represent a potential constraint on cropping, creating a restriction on crop treatment and a considerable inconvenience if they are managed according to regulations. Such inconveniences and potential yield reductions are acceptable within a low input system. In an era of reducing food stocks, requiring optimum production efficiency, they represent a serious restriction on production. However, they do provide opportunities to enhance the farm habitat and, if managed as uncropped grass based strips, allow the industry to make a positive contribution to rural biodiversity.

Acknowledgements
I am grateful to colleagues at Morley for helpful comment throughout preparation of this paper and also to R. Gobain of Stetchworth Estate Farms for suggestions made on an early draft.

REFERENCES

Anon (1991) *Code of good agricultural practice for the protection of water.* MAFF Publications, London SE99 7TP.
Anon (1995) *Guidelines for the management of field margins.* Fact sheet 2, The Game Conservancy Trust, Fordingbridge, Hampshire.
Anon (1996a) *Understanding buffer strips.* Environment Agency, Rio House, Aztec West, Bristol BS12 4UD.
Anon (1996b) In: Whitehead, R. (Ed) *The UK Pesticide Guide,* CAB International and BCPC. CABI, Oxford OX18 8DE.
Anon (1996c) *The Countryside Stewardship Scheme.* MAFF PB2505, London.
Fielder, A. (1988) Interactions between variety and sowing date for winter wheat and winter barley. *Research Review No. 6,* HGCA, London, N19 5PR.
Lotz, L.A.P., Wallinga, J.S. and Kropff (1995) Crop weed interactions: Quantification and Prediction. In: Glenn, D.M., Greaves, M.P. and Anderson, H.M. (Eds.) *Ecology and Integrated Farming Systems.* John Wiley & Sons, Chichester, PO19 1UD.
Mann, J.A., Harrington, R., Tatchell, M., Oakley, J.N., Jones, S., Ellis, S., Wothing, M. and Walters, K. (1995) Determination of factors affecting grain aphid movement with reference to spread by BYDV in the autumn and winter and forecasting direct damage in the summer. *Project Report 110.* Home Grown Cereals Authority, London NI9 5PR.
May, M.J. and Hilton, J.G. (1992) New spray techniques for broad-leaved weed control in sugar beet. *Aspects of Applied Biology,* **32**, 43-50.
Murphy, M. (1996). *Report on farming in the Eastern Counties of England 1994/95.* University of Cambridge, Department of Land Economy, Silver Street, Cambridge, CB3 9EP.
Nuttall, M. and Stevens, D.B. (1985) Winter wheat – Time of Sowing, 1982-1984. *77th Annual Report of the Norfolk Agricultural Station,* 67-70.
Nuttall, M. and Stevens, D.B. (1991) Winter wheat: Nitrogen and fungicide interactions in breadmaking wheat, experiment A – 1988-90. *83rd Annual Report.* Morley Research Centre, 81-96.

Oakley, J.N. (1994) Orange wheat blossom midge; a literature review and survey of the 1993 outbreak. *Research Review No. 28.* Home Grown Cereals Authority, London N19 5PR.
Polley, R.W. and Thomas, M.R. (1991) Surveys of diseases of winter wheat in England and Wales, 1979-1988, *Annals of Applied Biology,* **119**, 1-20.
Powlson, D. and Davies, B. (1993) Autumn and winter land management. In: *Solving the Nitrate Problem.* MAFF Publications, London SE99 7TP.
Smith, S.P., Davies, W.P., Bulman, C. and Evans, E.J. (1990) Responses of grain protein in winter wheat to nitrogen. *Aspects of Applied Biology,* **25**, 255-259.
Stevens, D.B. and Nuttall, M. (1993) Timing of reduced rates of nitrogen for winter wheat, 1990-1992. *85th Annual Report of the Morley Research Centre,* 158-168.
Stevens, D.B. and Yarham, D.J. (1981) Autumn diseases on cereals and their control. *Proceedings British Crop Protection Conference,* 725-738.
Stokes, D.T., Seath, R.K., Tilston, S.H., Cowie, G.S. and Sylvester Bradley, R. (1992) The effect of time of soil disturbance on nitrate mineralisation. *Aspects of Applied Biology,* **30**, 279-282.
Went, T. (1990) Response of *Bromus secalinus* and *B. sterilis* to pre and post emergence herbicide treatments. *Test of Agrochemical and Cultivations. Annals of Applied Biology,* **116**, Supplement 11, 68-69.

Buffer zones and farming systems

27

J. Baudry

National Institute for Agronomic Research, Sad-Armorique Research Unit, 65 rue de Saint-Brieuc, 35042 Rennes Cedex, France

Abstract
Agriculture is the main activity responsible for land cover/use change and non-point source pollution. It is important to study its effects from two different standpoints: how it affects buffer zones and how nutrient losses are related to buffer zones within landscapes.

This paper focuses on these two issues: 1) describing of the potential for buffer zones in different agricultural landscapes and systems and 2) analysing the spatial patterns of sources and sinks within these landscapes.

The paper is based upon data currently available in the scientific literature as well as upon conceptual models related to research on the potential for nitrogen control in farmed landscapes.

INTRODUCTION
Agriculture has been and still is the greatest cause of land use/land cover change (Turner II and Meyer, 1994). As such it has had a tremendous impact on buffer zones, in two different ways. The first is the transformation of riparian systems and, more generally, the conversion of wetlands into drained cropland. The second is the construction of hedgerows, grassy strips and ditches that may act as buffer zones.

The literature on the interactions between buffer zones and farming activities does not address directly the impact between the two, but develops two perspectives: the first is the efficiency of buffer systems with regards to the retention and transformation of pollutants and nutrients (Vought *et al.*, 1993); the second is the design and planning of buffer zones in agricultural landscapes (Van Buuren and Kerkstra, 1993; Haycock and Muscutt, 1995). The agronomic point of view often states that nitrogen is directly leached from the field to the water table (Sébillotte, 1992), which may be true in certain situations, as on limestone (Deffontaines *et al.*, 1993).

The objective of this chapter is to propose a framework and some hypotheses on the relationships between farming structure and function. We will also consider the type and fate of landscape elements that have a potential to act as buffer systems. We consider two such elements: riparian zones, that are stream corridors and which are potentially flooded for part of the year or at least have saturated soils, and hedgerows with associated ditches. Other chapters of this volume deal with the buffering potential of these elements. To illustrate our point we use studies on land use and farming systems in three French regions: Brittany, Normandy and Lorraine. Dairy production is dominant in all three areas, but the geomorphology, land use and level of production differ. The Normandy site (pays d'Auge) and the Lorraine sites are situated at the margins of the Bassin Parisien (west Normandy, east Lorraine), a sedimentary basin where soils have developed mainly on limestone and marl. The Brittany site (south of the Mont-Saint-Michel Bay) is on shale covered with loam.

THE FRAMEWORK: THE INTERACTIONS BETWEEN FARMING SYSTEMS AND LANDSCAPES
The farming system approach has been developed to analyse the interactions between farm components, namely the different types of production, farming techniques and field uses (Brossier

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0 © 1997 Quest Environmental.

et al., 1993). This concept is of overriding importance for understanding land use and landscape patterns and dynamics (Thenail and Baudry, 1994; Deffontaines *et al.*, 1995). This allows us to develop scenarios of change that can incorporate an environmental component (Ploeg, 1995), as opposed to a planned procedure alien to farming activities. From our standpoint, this approach allows us to understand how farmers use particular pieces of land within a specific type of production. The reason for differential uses may be the physical environment or work load but also farm territory (Josien *et al.*, 1994). The farm territory pattern also affects manure spreading (Le Houérou, 1994).

We have shown previously that the use of riparian zones can be explained by in-farm analysis (Thenail and Baudry, 1995). We make a distinction between the production system (type of production, capital, labour, etc.) and the technical system (technical means implemented to achieve the farmer or family goals). For instance, dairy cows can be fed from maize or grass silage or from grazing permanent grassland. Depending on the technique adopted, the impacts on the environment will be different.

The impact on riparian buffer zones can be a change in land cover (from grassland to cropland or woodland) or in land use (practices at field scale: fertilisation, hay-making, stocking rate, i.e., number of cattle head per ha). In general, more intensive techniques (higher inputs of nutrients, energy, labour, etc.) are used for fields grazed by dairy cows rather than heifers of beef cattle. Peyraud *et al.* (1995) have shown a positive relationship between the herbage production, the stocking rate and the quantity of nitrogen excretion by dairy cows. In a simulation study, Dijkstra and Hack-ten Broeke (1995) showed that nitrogen leaching varies according to grazing management and soil type.

We extend our analysis to gain a general insight of the factors affecting riparian zone cover and use, as well as the maintenance of hedgerows.

THE HYPOTHESES

As riparian zones have more constraints, from an agronomic point of view, they will be used for less demanding production. However, there is a diversity of situations which we can relate to:

1) the farm territory (size, location). The larger the relative area of riparian zones within a farm, the more it will be intensively used. In dairy farms, the closer riparian areas are to the farmstead the more they will be used intensively for feeding the cows that are brought back to the barns for milking.
2) the type of production and associated technical systems. Some types of production systems (cash crop) or technical systems (feeding animal with maize instead of grass) require that all land be ploughed to be used, otherwise it will be abandoned.

The variables we use are: total farm area, area of riparian zone, type of farming system and, for stock farms, type of grazing animal.

In regards to hedgerows, the dominant view is that intensification is correlated with hedgerow removal.

THE LORRAINE CASE STUDY

Benoit (1985) studied the land use of the fields of fifteen neighbouring stock farms where milk was the main production. Half of the farms had fields on alluvial soil; that is 20% of the area. For farms with alluvial soils, the area of alluvial soils was proportional to the total farm area. The use of the alluvial soils varies from farm to farm. Only three farms out of seven used part of their alluvial land for grazing dairy cows (Fig. 1a), while all of them used part of this land for grass silage production (Fig. 1b). We can consider that dairy cow grazing creates a high risk of pollution from excreta, while silage removes biomass from the fields. The main reason for the differences in usage is distance from farmstead; for alluvial soils it follows the general trend (Table 1), except for sheep grazing. The relative location of alluvial soils and barns influences the use of these fields.

Figure 1. Lorraine: a) riparian zones and farm territory, b) use of riparian zones

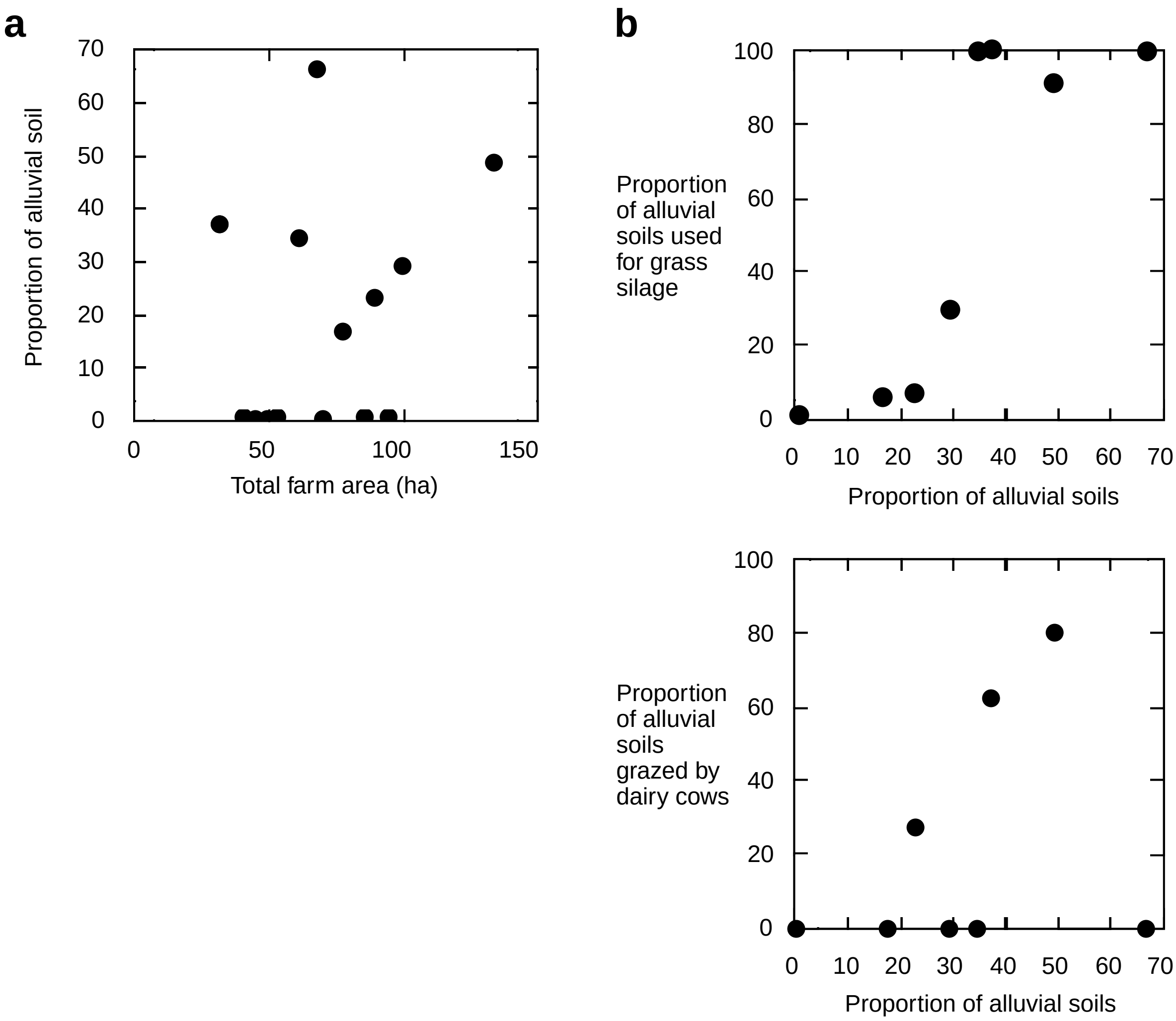


Table 1. Lorraine; relationship between land use and distance to farmstead for non-alluvial and **alluvial** soils, in a number of fields

	Dairy cows	*Forage & cows*	*Forage & heifers*	*Heifers*	*Silage*	*Hay*	*Crop*	*Sheep grazing*
0-100 m	6	1	0	0	3	1	1	1
	10	**0**	**0**	**0**	**0**	**0**	**0**	**0**
100- 500 m	16	4	0	1	1	0	3	5
	2	**3**	**1**	**0**	**1**	**1**	**1**	**5**
500- 1000 m	8	17	0	3	4	4	3	35
	0	**1**	**3**	**2**	**4**	**2**	**9**	**7**
1000- 1500 m	6	6	0	2	12	6	9	47
	0	**0**	**0**	**1**	**7**	**0**	**4**	**3**
1500- 2000 m	1	3	0	2	5	8	11	30
	0	**0**	**0**	**0**	**1**	**1**	**1**	**1**
>2000 m	0	0	0	2	13	4	30	28
	0	**0**	**0**	**0**	**2**	**0**	**7**	**1**

THE PAYS d'AUGE CASE STUDY

This area of central Normandy is mainly covered by permanent grassland (almost 80% of farmland). The substrate is very heterogeneous, changing from limestone to marl and to alluvial soil. Stock farm types are diverse; specialisation in dairy to beef cattle production occurs and recent developments have occurred with the arrival of mixed farming with beef and cereal production (Baudry *et al.*, 1993; Thenail and Baudry, 1994). The use of riparian zones varies from light grazing on unfertilised pasture to heavy grazing or ploughing of the zone. We have assessed the actual land use in a 800 ha municipality/parish where land use is known for each field. The municipality is traversed by a stream and there are several springs and associated wet zones; these coincide with the base of limestone and the top of the marl geology.

The riparian zones also form around 20% of the total land area. In this municipality, over 90% is under permanent grassland, less than 2% of the land is ploughed. 58% receives no fertiliser. We classified the farming systems along a gradient of intensification from type 1 that has 100% permanent grassland to type 5 where all arable land is ploughed (fields that cannot be ploughed are under permanent pasture).

When related to farm type, it appears that even though grasslands in the riparian field are more frequently unfertilised than others (Table 2), in farm types with dairy specialisation they receive some nitrogen. In the most intensive farms, intensification through the production of forage crops (maize) takes place on mesic soils, while the riparian zones are grazed by heifers or beef cattle. Both the type of production system (the farm product) and the technical systems (means of production) drive the use of the riparian environment.

Table 2. Pays d'Auge; permanent grassland fertilisation and farming systems (% pasture per farm type)

Type of farming system	*No fertilisation*	*Approx. 50 kg N per annum*	*Approx. 100 kg N per annum*
Small 100% grass	82	5	13
Less intensive dairy	44	56	0
More intensive dairy	54	0	46
Cash crops/ beef	100	0	0

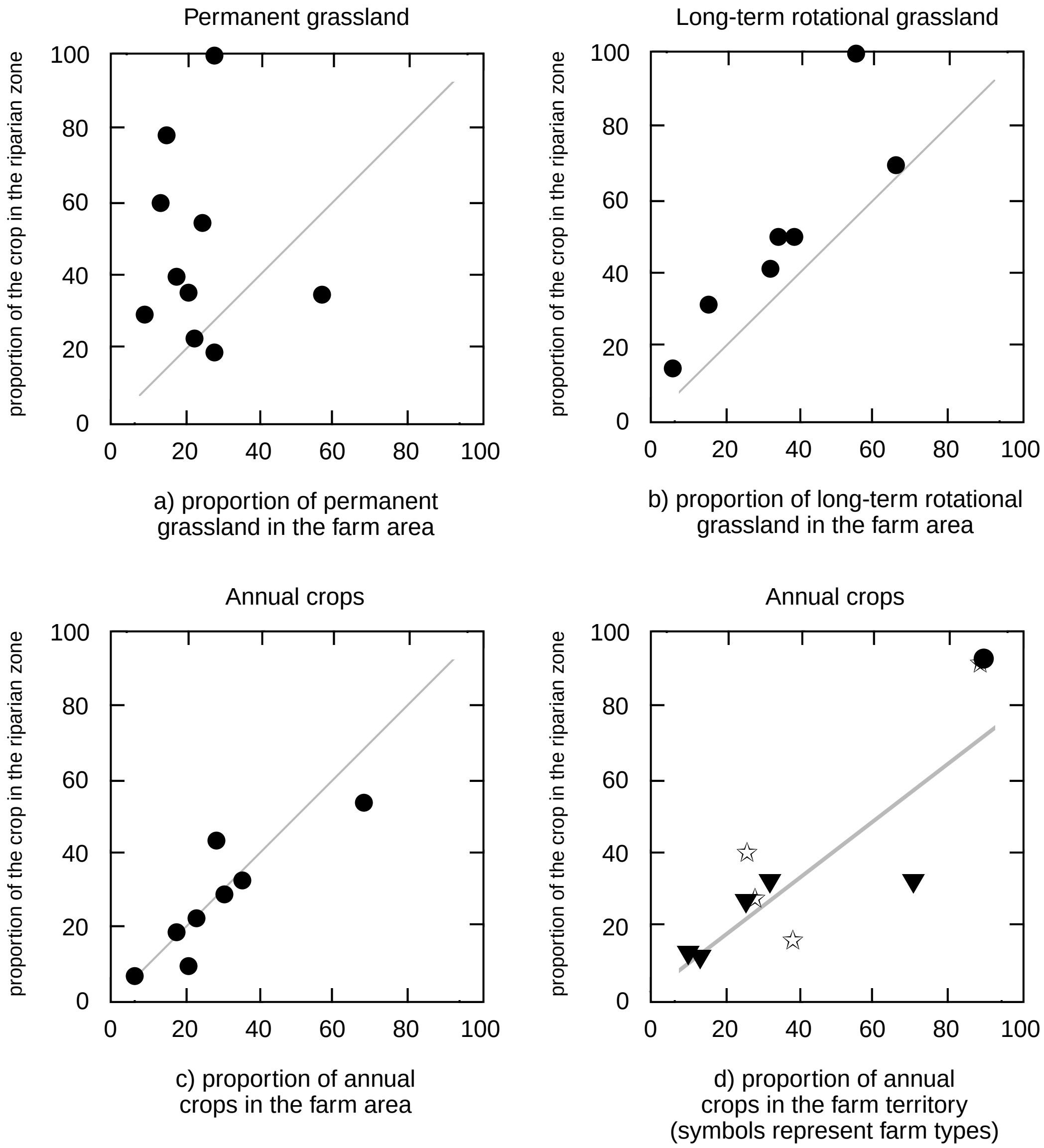
NORTHERN BRITTANY

In northern Brittany we studied a 700 ha landscape traversed by three brooks (Burel, 1996; Thenail, 1996). The substratum is shale covered by up to 60 cm of loamy soils. The riparian zones (delineated as hydromorphic soils) cover 30% of the area. Farms are predominantly dairy farms, with some farms producing cash crops (wheat). We consider the three dominant types of land cover: permanent pasture, long term rotational grassland and annual crops (wheat, maize) associated with dairy production. Overall, permanent grassland represents 46% of the riparian zones, woodland and old fields about 14%. Sixty-five percent of the permanent grassland is associated with the riparian zones.

Figure 2 shows that, at the farm scale, permanent grassland is found more frequently in the riparian zones than over the farm territory as a whole, long-term rotational grassland is proportional to the overall frequency. For annual crops the frequencies are proportional, though some farms have less crops in the riparian zones. The figure also shows that the more riparian zones a farm has, the more these are used for annual crops; this is not the case for permanent or long-term grassland.

Land cover is related to farm type (table 3). Cash crop farms plough all the land and the amount of permanent grassland decreases as the milk quota increases. In parallel, either rotational grassland or annual crop coverage increases. We did not find any effect of distance on land cover, whatever the farming system.

Figure 2. Brittany; proportion of crops within the riparian zone against the proportion in the total farm area (a-d)



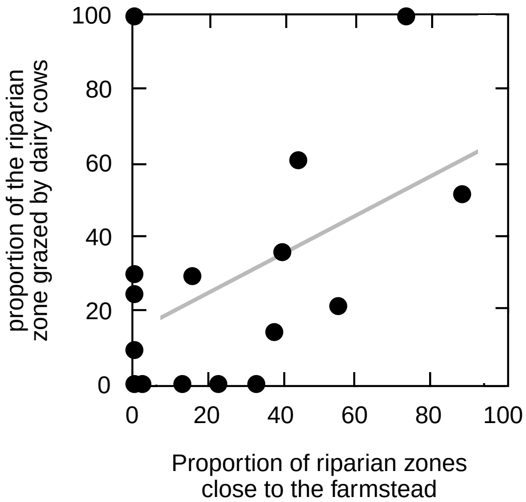
This trend is shown by the relationships between land cover and type of farming system (Table 3). Cash crop farms plough all the land and the amount of permanent grassland decreases as the milk quota increases. In parallel, either rotational grassland or annual crop coverage increases. We did not find any effect of distance on land cover, whatever the farming system.

Table 3. Brittany; use of riparian zones and farming systems (% fields per farm type; not all production types were considered).

Type of farming system	*Permanent grassland*	*Rotational grassland*	*Annual crops*	*Woodland old fields*
Farm stopping milk production	50	5	0	0
Low quotas, milk	52	9	12	10
Low quotas, mixed	47	9	14	12
High quotas, milk	28	14	10	42
High quotas, mixed	57	42	0	0
Cash crops	0	0	66	0

At a farm scale, a strong relationship appears between the proportion of riparian zones grazed by dairy cows and the proportion of riparian zones near to (positive relationship) the farmstead (Fig. 3). The trend is reversed for the other types of animals that usually graze at a lower stocking rate.

Figure. 3. Brittany; frequency of riparian zones grazed by dairy cows as a function of the distance from farm buildings



HEDGEROWS ON THE BRITTANY SITE

Only in the Brittany study sites were we able to relate hedgerows to fields and farms. We did not make any distinction between hedgerows as we do not have any criteria concerning their buffering potential. As it is difficult to know to which farmer a hedgerow belongs, we considered the fields and their surrounding hedgerows; we attributed to each farmer the length of hedgerows bordering his fields. Therefore a hedgerow between the fields of two different farmers was counted twice. This method allows the assessment of the density of hedgerows per farm and per farm type, but the figures are meaningless at the landscape scale.

The density of hedgerows depends on the farmers' willingness to maintain them between his fields, but also of the ownership patterns: scattered isolated fields are more often surrounded by hedgerows. Table 4 shows that the most productive farm types are associated with a lower hedgerow density. It is worth noting that, at the same time, these most intensive farms have the highest proportion of hedgerows between two adjacent fields of the same farm, except for the cash crop farms. Thus the less intensive farms have a more dispersed territory, and intensive farmers appear not to systematically remove their hedgerows.

Table 4. Brittany; type of farming systems and hedgerow density

Type of farming system	*Hedgerow density ($m\ ha^{-1}$)*	*% Hedgerow length between two fields on same farm*	*Density of hedgerow between two fields on same farm*
Farm stopping milk production	368	13	63
Low quotas, milk	399	12	48
Low quotas, mixed	448	17	75
High quotas, milk	279	21	60
High quotas, mixed	220	20	45
Cash crops	368	8	30

DISCUSSION

As far as we know, no research programme has been designed yet to evaluate the interactions between farming systems and the landscape elements that can act (at least potentially) as buffers against nitrogen fluxes at the landscape level. The use of available data from surveys of land cover/use in stock farms has yielded promising results. Some patterns have emerged that will help us to design further surveys.

For riparian zones, several variables at different levels influence the land use patterns:

- at a regional scale, a large quantity of riparian land tends to be permanent pasture.
- the farming production type influences the requirement for land. This has an impact on riparian land use.
- at the farm scale, the relative distance of the riparian zones to farm buildings affects the local use of the riparian zones, particularly with regards to grazing pressure.

For hedgerows, both field scale use and farm type are related to the presence and structure of these field boundaries.

Two approaches are possible to understand buffer zones uses. The first is to start from landscape maps and to correlate the field's use with farming patterns (type of production, distances, etc.). The second is to start from models of farm functioning, specifically decisions on land use (fertilisation, stocking rate, etc.). The first would be suitable for a broad scale approach, involving a concise survey of many farms, while the second can only be done in a limited number of farms. We need both to link the spatial patterns and mechanisms at the farm scale.

The data we used allow the detection of trends but do not permit statistical tests. The main reason is that there is a need for a larger number of farms to be integrated to achieve a diversity of farm types at the various levels. One other problem is that it is not possible to survey all the farms on a field-by-field basis in a given landscape; thus our farm data (except for Lorraine) are not complete. On the other hand for Normandy and Brittany, we have no data on the type of soils within fields of the surveyed farms. This is certainly the major problem in linking landscape patterns and farming systems. Running simulation models and testing the outcomes against survey data is a further way to improve our knowledge of the patterns of riparian land use.

Even our preliminary findings are important from an applied point of view. They imply that policies regarding buffers must be designed within a landscape and a farming system context, taking into account areas outside the riparian zones. The protection and restoration of buffer zones may require profound changes in the technical functioning or the type of production of the farms of the area. One would think that these changes will be more important for riparian zones than for hedgerows. The latter can be fitted around the fields in most farm types, while the former, being a resource base for production, has to fit in to the process of farm production. These results also point to the necessity of considering both the effects of land cover and land utilisation on the functioning of the buffer zones.

If a farming system perspective is useful for the design and management of buffer zones, it is also true that a buffer management perspective is useful for farm management. The latter is often planned at the field scale, with no consideration of spatial patterns, or impact, at the landscape scale (Knickel, 1994).

ACKNOWLEDGEMENTS

I warmly thank Claudine Thenail for providing the data from Brittany and Marc Benoît for those from Lorraine. I am grateful to Françoise Burel and Claudine Thenail for their comments on an earlier version. Thanks also go to the Ministry of Environment (Comité Ecologie et Gestion du Patrimoine Naturel) for providing funding through its programmes on Agriculture and Environment. Thanks also to the editors for their comments and corrections to the text.

REFERENCES

Baudry, J., Thenail, C., Le Coeur, D., Burel, F. and Alard, D. (1993) Landscape ecology and grassland conservation. In: Haggar, R.J. and Peel, S. (Eds.) *Grassland management and nature conservation. Occasional Symposium n° 28. British Grassland Society,* pp. 157-166.

Benoît, M. (1985) *La gestion territoriale des activités agricoles, l'exploitation et le village : deux échelles d'analyse en zone d'élevage.* Cas de la Lorraine (région de Neufchateau), Institut National Agronomique Paris-Grignon.

Brossier, J., de Bonneval, L. and Landais, E. (1993) System studies in agriculture and rural development. *Science Update,* pp. 415. INRA Editions, Paris.

Burel, F. (1996) Hedgerows and their role in agricultural landscapes. *Critical Review in Plant Sciences,* **15**, 169-190.

Deffontaines, J.P., Brossier, J., Benoît, M., Chia, E., Gras, F. and Roux, M. (1993) Agricultural practices and water quality. A research development project. In: Brossier, J., De Bonneval, L. and Landais, E. (Eds.) *Systems studies in agriculture and rural development,* pp. 31-61. INRA.

Deffontaines, J.P., Thenail, C. and Baudry, J. (1995) Agricultural systems and landscape patterns: how can we build a relationship? *Landscape and Urban Planning,* **31**, 3-10.

Dijkstra, J.P. and Hack-ten Broeke, M.J.D. (1995) Simulation of different management options within integrated arable farming affecting nitrate leaching. In: Schoute, J.F., Finke, P.A., Veenklaas, F.R. and Wolfert, H.P. (Eds.) *Scenario studies for the rural environment. Environment & Policy,* pp. 329-333. Kluwer Academic Publisher, Dordrecht.

Haycock, N.E. and Muscutt, A.D. (1995) Landscape management strategies for the control of diffuse pollution. *Landscape and Urban Planning,* **31**, 313-322.

Josien, E., Dedieu, B. and Chassaing, C. (1994) Etude de l'utilisation du territoire en élevage herbager. L'exemple du réseau extensif bovin Limousin. *Fourrages,* **138**, 115-134.

Knickel, K.W. (1994) MSSBB – A concept and microcomputer application for agro-environvironmental land use analysis and planning. In: Fresco, L.O., Stroosnijder, L., Bouma, J. and van Keulen, H. (Eds.) *The future of the land: Mobilising and integrating knowledge for land use option,* pp. 337-344. Wiley & Sons, West Sussex.

Le Houérou, B. (1994) Transfert du fumier des cultures vers les prairies en Lorraine. Outils et méthode pour changer de pratiques. *Fourrages,* **140**, 471-488.

Peyraud, J.L., Vérité, R. and Delaby, L. (1995) Rejets azotés chez la vache laitière : effets du type d'alimentation et du niveau de production des animaux. *Fourrages,* **142**, 131-144.

Ploeg, V.D. (1995) The tragedy of spatial planning. In: Schoute, J.F., Finke, P.A., Veenklaas, F.R. and Wolfert, H.P. (Eds.) *Scenario studies for the rural environment. Environment & Policy,* pp. 75-90. Kluwer Academic Publisher, Dordrecht.

Sébillotte, J. (1992) Pollution des eaux par les nitrates : questions sur les prairies. *Fourrages,* **132**, 347-354.

Thenail, C. (1996) *Exploitations agricoles et territoire(s) : contribution à la structuration de la mosaïque paysagère.* Thèse de doctorat thesis, Université de Rennes 1.

Thenail, C. and Baudry, J. (1994). Méthodes d'étude des relations entre activités agricoles et paysages. In: *Recherches-Systèmes en Agriculture et Développement Rural. 13e Symposium International,* pp. 316-321. CIRAD.

Thenail, C. and Baudry, J. (in press). Consequences on landscape pattern of within farm mechanisms of land use changes (example in western France). In: *Land use changes in Europe and its ecological consequences. (IALE seminar).* Wageningen Agricultural Press, Tilburg.

Turner II, B.L. and Meyer, W.B. (1994) Global land-use and land-cover change: an overview. In: Meyer, W.B. and Turner II, B.L. (Eds.) *Changes in land use and land cover: a global perspective,* pp. 3-10. Cambridge University Press, Cambridge.

Van Buuren, M. and Kerkstra, K. (1993) The framework concept and the hydrological landscape structure: a new perspective in the design of multifunctional landscapes. In: Vos, C.C. and Opdam, P. (Eds.) *Landscape ecology of a stressed environment. IALE Studies in landscape Ecology,* pp. 219-243. Chapman & Hall, Cambridge.

Vought, L.B.-M., Pinay, G., Fuglsang, A. and Ruffinoni, C. (1993) Structure and function of buffer strips from a water quality perspective in agricultural landscapes. *Landscape and Urban Planning,* **31**, 323-332.

Integrating vegetative buffer zones within catchment management plans

28

J.L. Gardiner and C. Perala-Gardiner

School of Geography and Environmental Management, Middlesex University, Queensway, Enfield, London, EN3 4SF, UK and Salix International Ltd. UK.

Abstract

Land use determines the quantity and quality of the water environment, which is not only vital for the natural environment, but in turn underpins the use of the land, whether for agriculture, urban development or recreation. Management of the relationship between land use and the water environment is therefore critical, although it is subject to the cumulative impact argument which allows insidious decline in so many ways.

Catchment Management Plans (CMP) can potentiate a synthesis between what have hitherto been largely regarded as separate fields of interest; they can therefore be seen as fundamental to achievement of sustainable use and development of the land. Changing the worldview of water as an exploitable, infinite resource or as a waste product into a valuable resource to be husbanded with the greatest care is a daunting challenge assisted greatly by periods of drought.

There is a strengthening movement in the UK towards source control (changing the surface water disposal into a surface water management paradigm) which promises to prevent low-flow or pollution symptoms downstream. However, this will take effect over many years, and most rivers need protection now. Buffer zones can achieve this intermediate protection and catchment analysis can indicate where such zones are most required to benefit ecology, water quality, fisheries, recreation etc.. There are direct analogies with the zoning for land associated with water resources and the identification of statutory water quality objectives (SWQOs).

For catchment management plans to play a proactive role in determining where buffer zones are most required, the approach to CMPs needs review. At present it has been seen as a quick job, to put together all the Environment Agencies functions between one set of covers – and that is indeed a good start. However, the process itself can add considerably greater value to the product, by requiring the CMP teams to investigate a series of generic issues such as the need for buffer zones. Then the Environment Agency can truly say it has made a lasting difference in terms of multi-functional support of sustainable development.

INTRODUCTION

This paper sets out to review and progress our understanding of the crucial role of vegetative buffer zones within river catchments and the need to integrate these within river catchment management plans. We emphasise the need to enhance natural processes to support the ecological, hydrological and geomorphological continuity of the catchment. Powerful enhancements flow from structural integration of vegetation with engineering (called "soil bioengineering"), within plans for development. Such structural uses of vegetation have particular application on the banks of watercourses, in uplands for bank stabilisation and gully erosion and to manage the water quality of runoff from the hinterlands. These concepts are derived from three approaches having worldwide currency: environmentally sensitive river engineering, catchment management planning and working with rather than opposing natural processes.

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0 © 1997 Quest Environmental.

Integration of multiple functions in the buffer zone

Many other facets of development point in the same direction, such as the need to seek low-cost prevention rather than high-cost cure. Buffer zones at the source of urban runoff (for example in the form of grass swales) or in the riparian zone, can reduce both the use of manufactured materials and expensive on-site processes to achieve waste minimisation and greater flexibility in design. Vegetated buffers will allow future generations to apply adaptive management at least cost. All these benefits are parts of the drive towards sustainable development, including both the biodiversity protocol and Agenda 21 itself. We define a buffer zone in this paper to include any vegetated area interposed between the point of rainfall and the primary river channel, which acts to mitigate the influence of land use (including in-stream activities) on the riverine environment.

This subject draws together many issues within the catchment landscape. Increasing awareness of the complex issues of land uses and their effects on the water environment implies a greater appreciation of the connections between the individual site, the locality, the surface water drainage system, the river reach and regional processes, governed by climate, geology, land use and vegetation cover at a catchment scale. Most important are the action steps needed to change assumptions and actions in development strategies and land management behaviour.

CATCHMENT ASSESSMENT FOR POTENTIAL ENVIRONMENTAL ENHANCEMENT

Several physical attributes are unique to each catchment, such as geology and geomorphology, response to rainfall, ecology and patterns of development or land use (Gregory and Walling, 1973). All these are interdependent, but the pattern of land development is often the most significant determinant of change in these other features.

Most land use changes under the traditional title of "development" tend to destabilise the existing equilibrium between soil, flora and precipitation both locally and, with cumulative impacts, regionally. The direct effects of urbanisation and of such practices as forest clearcutting, overgrazing and of converting floodplain pasture into arable land are reasonably well understood (Newson, 1992). Arguments continue over the need for such exploitative activities aimed at economic growth rather than husbandry of resources and the need to retain the habitats and species so endangered. A strong case, based on an economics which includes environmental costs, can be made not only for mitigating all environmental damage from the development as much as possible, but also for recovering from inherited damage, through environmental enhancement (Gardiner, 1988). Such enhancement may be more than what has been commonly regarded as economically justifiable, but can be seen as an example of applying the precautionary principle within the sustainability approach (Pearce, 1995, p.19).

We argue that, to be sustainable, any new development must conserve, enhance or reinstate those natural attributes of a catchment which support its ecological functions and which may well have made development initially possible or desirable. Given the ability of the river corridor to reflect the health of the catchment, overall success criteria could include the relative stability of riverbanks, good water quality, retaining natural thresholds of flooding and environmentally acceptable minimum flows.

Development and sustainability

Since the initial "development" of a natural catchment cannot, by definition, proceed with full mitigation of every impact on the natural systems, some "balance" has to be achieved, relying on redundancies in the resilience of those systems to ensure overall sustainability. What are these redundancies? How can we tell when they are about to disappear and when we are at the threshold of the "carrying capacity" of the reach or the catchment, beyond which we are in danger of losing "critical natural capital"?

Adequate knowledge of this seems impossible to achieve; every catchment under development has a legacy of environmental damage, which includes degraded river corridors. Some areas can be restored to a condition resembling a former state, or rehabilitated, whether by design or purposeful neglect, to some state or process agreed as appropriate to the changed circumstances. The implication is that there is nearly always massive scope for enhancement and the precautionary principle would suggest that it is further development, rather than growth, which should support this enhancement (Lovins and Kinsley, 1995).

The broader community and need for cooperation

The potential for enhancement can be realised by cooperative efforts in the community, especially among local authorities, infrastructure management and maintenance authorities and the development industry. An example is the conservation of the endangered El Segundo Blue butterfly and its habitat beside Los Angeles International Airport (LAX). This effort was coordinated among the City of Los Angeles, LAX, Exxon Petroleum, the California Native Plant Society and a host of local environmental groups and concerned biologists. Many other examples are found in the work of the Society for Ecological Restoration based in Madison, Wisconsin, USA; the Urban Streams Restoration Program in California and the Australian Landcare programme, as described below, and the River Medway Restoration Programme in Kent, England and elsewhere.

One of the more obvious benefits of such cooperative efforts is the decreased likelihood of one organisation acting solely in its own interests – a major cause of environmental destruction associated with single-function decision-making. Economic costs and benefits distributed equitably among the stakeholders would ensure that (re-) developments bear their share of funding an associated programme of environmental enhancement, through a systematic analysis of the opportunities and community goals. The issue of timing for the shared funding of these enhancements can be significant, but not insuperable, if local authorities can facilitate equitable distribution of the responsibilities and benefits.

Geographic data integration

Geographical information systems (GIS) are powerful tools for technical, educational and political purposes. As a spatial reference map, GIS can be used to guide the assessment of the pattern of potential environmental enhancement for a catchment, guided by a strong indication of local and regional priorities. A multi-dimensional map and database are essential for a monitoring programme, to chart progress toward some agreed target of sustainability. The question remains: "How to prioritise?". This needs information and a method of decision-making which incorporates stakeholder views, which leads in turn to catchment management planning as a tool for decision-making among stakeholders.

CATCHMENT MANAGEMENT PLANNING AND STRATEGIC ENVIRONMENTAL ASSESSMENT

There are several versions of catchment planning practised round the world, ranging from Europe, the USA and South Africa, to Malaysia, Hong Kong and Australasia (and many other countries en route). Words such as "integrated" and "total" commonly fail in practice to include all the significant players in land use and the water environment, usually because the institutional arrangements do not facilitate such coordination. New Zealand can claim institutional integration, with Regional Councils, set up in response to the Resources Act of 1991, combining local authorities with the 40-year-old Soil and Water Conservation Authorities to cover all aspects of land use planning and water management.

Elsewhere, it is the *holistic approach to catchment planning* which can coordinate the activity plans of local (land use) planning authorities with plans for water resources, in the global sense of the phrase (Gardiner, 1990). Local planning bodies were charged at the 1992 Earth Summit at Rio de Janeiro with delivering the substance of Agenda 21. The term "water resource planning agencies" implies the entire range of people who manage the water environment, from flood defence through water quality, supply, sewerage, navigation, fisheries, conservation and recreation. In the UK, the newly-formed Environment Agency has added the waste regulation duties of local authorities and the air quality duties of the former HMIP (Her Majesty's Inspector of Pollution) and other pollution control duties of the former HMIP, to the former National Rivers Authority (NRA).

SUPPORTING TECHNIQUES

The process of plan coordination uses data and information about catchment attributes, uses and bio-geophysical/chemical processes. The topography needs to be related to the pattern of land use and ownership, for which an important source of information now comes from satellites as well as aerial photographs. Such remote sensing allows not only quantitative information (for example, on the extent of riparian tree cover) to be gathered but also qualitative information (for example, on the health of the trees) to be assessed. Comparison of images over time (historical analysis) allows assessment of geomorphological change, particularly in terms of river movements. The extent of land use change can readily be seen in an historical analysis and the synoptic view is often surprising as one realises just how rapid recent change has been. Again, a GIS type strategy can be a simple and powerful means of integrating these data.

River modelling

An economically justifiable and comprehensive assessment of the flood risk and the effect of floodplain storage can be achieved by modelling the river system in stages. A first stage will model the catchment's hydrological response, a second stage will model the river system and, where flooding is an issue, a third stage might be to create a digital ground model, based initially on aerial survey of the floodplain. Ground surveys, including topography, river habitat, landscape and geomorphology, are vital to provide further detail to aerial survey when change to the river and its floodplain is being considered.

Strategic environmental assessment (SEA)

The planning process does not only rely on technical support. Whatever the nature of the plans to be coordinated (to produce effective "integrated catchment planning"), coordination requires a procedure to bring the plans together. Strategic environmental assessment (SEA) offers that procedure (Therivel, 1994), ensuring that stakeholders are involved in a process which fully supports the holistic approach to planning (Gardiner, 1996). A significant step towards such a regional approach has been made in the UK with "Thames 21" (NRATR, 1995). This document articulates a planning perspective and a sustainable strategy for the Thames Region, a highly-developed catchment of some 13,000 sq km supporting a population of some 12 million people. The aim is to involve the stakeholders to the extent that ownership of the *process,* as well as the resulting activities, is shared with them.

The conclusion of this brief overview is that there is sufficient experience and understanding of the institutional processes and techniques needed to enable vegetated buffer zones to be established. The physical buffer zone is a direct means of both mitigating the impacts of land use and also enhancing or reinstating the functionality, if not some of the redundancies, of the catchment's ecosystems. But how well do we understand the potential for such buffer zones? Where can they be installed and effectively maintained over the long term?

POTENTIAL ROLES FOR REVEGETATION ENHANCEMENT: SOIL BIOENGINEERING

Past understanding of landscape vegetation cover has been based on either aesthetics or wildlife habitat. We offer a third class of vegetation use: soil bioengineering, which integrates plant ecology and engineering (Schiechtl, 1980). This is a structural approach to the use of both woody and herbaceous plants, which offers integrated benefits due to the physical properties of plants in a landscape. The following is a summary of the categories of structural vegetation uses from a catchment perspective. These categories relate to the recovery of ecosystem functions, as these functions benefit human demands on natural resources.

Vegetation in created wetlands for water quality

Many methods now exist for treatment of water pollution using constructed wetlands (Hall *et al.*, 1993). An overall theme of these approaches may be summarised by increasing the contact time between surface waters, plants, air and soils. Adequate retention of the polluted water improves water quality by various means, all requiring incorporation of oxygen into nutrient-rich waters. The more passive the system (i.e. the fewer the energy inputs), the more land is required, but this can often be accomplished using marginal lands, as was the case with an early example of this strategy, the coastal municipal sewage treatment marsh at Arcata, on the west coast of California.

Many such water treatment schemes have been developed in Europe and the United States. Most of these feature the use of plants, especially riparian trees and the sedges, rushes and grasses, to provide the aquatic substrate on which functional algae and bacteria (microflora) grow (Lowrance *et al.*, 1995). While these plants do absorb pollutants in their root systems, stems and leaves, it is often the microflora which do much of the biochemical work of nutrient conversion. Vegetated, created wetlands can provide multiple benefits such as flood water storage, wildlife habitat, fibre extraction, open space for passive recreation and other amenities. These multiple functions provide a broader range of benefits, hence increasing the potential benefit-cost ratio over the achievement of these aims by other means, making wetland treatment of water more economically feasible than single purpose treatment.

Vegetation for storage ponds and source control

The use of storage (or attenuation) ponds and stilling basins is becoming more commonplace in Europe. These created ponds are often used, for example, to collect road runoff and thereby reduce peak storm flows. Attenuation ponds can also be used adjacent to parking lots, to collect roof runoff, or in industrial parks to collect rainwater. Only recently has purposeful design for water quality been included with the hydrological, wildlife and recreational functions (Hall *et al.*, 1993). This approach focuses on integrating vegetation as structural elements in the built environment to achieve multiple functions within the river corridor, such as:

1. to direct or distribute flow (e.g. to ensure an even flow through the lake and minimise the development of channel incision)
2. to attenuate flow and thereby achieve the design contact time
3. to reduce flow velocities
4. trap suspended sediments (and associated pollutants)

The major difference between attenuation ponds and source control techniques may be found in the way the use of vegetation is combined with the urban landscape. Source control is the control of runoff at or near the point of rainfall. An example of this is the use of grass/reed swales to clean storm water runoff from housing developments, as seen in Reston, Virginia, USA. We still face a considerable institutional and technical challenge to integrate urban planning with design for the water environment. Where the public calls for a more "natural-looking" environment, there is more opportunity for structural use of vegetation to replace hard elements of the built environment.

Such strategies offer opportunities for creative soil bioengineering design, using material ranging from porous pavements, turf and grass grids, geotextiles, the grasslike sedges and rushes, to willows and other wet-tolerant woody shrubs and trees. Porous paving stones under a roof outfall can be vegetated with rushes (*Juncus* spp.) and sedges (*Carex* spp.) to integrate rainfall infiltration, stormwater attenuation, soil erosion reduction, water quality improvement and microhabitat for beneficial insects. Such a change may require new approaches to maintenance, but does not necessarily imply significant increase in overall maintenance requirements and may in some circumstances reduce maintenance costs.

Vegetation in upland buffer zones for soil stabilisation

From the catchment perspective, control of upland erosion for sediment and water retention is more cost-effective than trying to improve sediment imbalances in the downstream river channel. If the health of the riverine environment is a goal, e.g. for recovery of salmonid fish habitat, it usually makes more sense to apply erosion control solutions first in upland areas, to the sources of eroding fine and coarse sediments. However, where instream processes are driving headward recession, revegetation of river banks using the principles of soil bioengineering may be needed, beginning downstream and working in the upstream direction (Roseboom, 1996, pers. comm.). Upland rill and gully erosion express the causes of aggravated soil loss; correct problem identification is the key to reducing downstream sedimentation. Two of the most common factors in upland erosion are:

- loss of vegetation from factors such as overgrazing, conversion of pasture to arable land, logging and construction
- recent changes in surface hydrology and compaction of soils. Where storm flows are directed at soft soils, vulnerable surfaces experience high shear stress. Vehicles disrupt and compact the soil structure. Examples of this include large-scale excavation works, road construction, inappropriate placement of culverts, off-road vehicles and logging transport on forest "roads".

Although less obvious and less glamorous than downstream projects, control of upland erosion is vital to the recovery of geomorphological equilibrium at the catchment scale. Regional assessment of upland problems should use the stream network to identify those areas which have the greatest erosive potentials; steep slopes and the margins of existing upland channels (Gregory and Walling, 1973). Many upland erosion control techniques have been identified, by Schiechtl (1982), Gray and Sotir (1995) and others.

Vegetation for riparian margins and floodplains

Bringing vegetation into river management first requires flexible thinking and a willingness to give More Room for Nature, in German "Mehr Raum fur die Natur", a phrase describing the basic element of the "Leitbilt", the vision which gives direction to current river restoration in Germany (DWVK, 1996). The principles of fluvial geomorphology offer a sound basis for river management, using naturally-occurring processes in these dynamic environments. The direction of river management is now toward integration of the river channel with its floodplain. The principles of fluvial geomorphology show that allowing wider floodplains free from permanent development can repay us richly in safety from hazards and diverse landscapes for our benefit as well as that of other species. Soil bioengineering used in riverbank stabilisation has demonstrated that working with inherent river process is less costly than trying to command the flood.

More land given to river processes will repay us economically with multiple functions and benefits integrated into the floodplain land use. A brief inventory of these functions includes affordable flood defence, water quality improvements, groundwater recharge, increased water supply, provision for the increasing demand for recreation near settlement, aesthetic landscapes, wildlife habitat, safe fisheries, navigation, irrigation, food and fibre extraction, mineral resource extraction and so on.

However, the soil bioengineering profession may be in danger of attempting to substitute trees for rock rip-rap by excessive training of river banks; bank erosion is vital to the health of every river system. The tools of geography and geomorphology can aid the process of asking the relevant questions, such as: Where are bank protection structures needed? Where can the river be allowed to shape itself? How much intervention is needed? What opportunities does the catchment provide? Land use planning is the first tool of ecological engineering, so that precious resources are not spent protecting lands better left to the flood.

A diversity of geomorphological features in the river landscape leads toward a connected river corridor and toward lateral connection between main channel and floodplains. The design of stream bank reinforcement using flexible vegetation structures must allow for stream hydrology, sediment transport and deposition regimes, ecological criteria and land uses (Perala, in press). Soil bioengineering provides a key focus for the multi-disciplinary team in their efforts to recover functional rivers and their floodplain wetlands. Many ecological approaches to river management techniques have now been articulated (RSPB *et al.*, 1994) and river engineers will need to rely on the experience and training of the "ecological engineer" to assess which method is most appropriate for any given situation.

GUIDANCE FOR DESIGNING THE SITE-SPECIFIC TECHNIQUE

Specific erosion control techniques must be evaluated for the catchment and designed for the particular site. The greatest challenge in upland erosion control is correct identification of the problem.

The option of "benign neglect" or "allowed natural recovery" should always be considered first, allowing vegetation to recover by removal of the problem, such as excessive herbivore from grazing cattle. A substantial amount of the excessive sedimentation in Western American stream channels at the present time is being corrected by reducing grazing pressure on upland ecosystems, evidenced by a policy shift by the USDI Bureau of Land Management (Prichard, 1996, pers. comm.). Where erosion is primarily of surficial origin, revegetation may include structural elements such as fascine bundles or siltation baffles. These structural methods should be used to foster the *in situ* soil-forming process and to initiate the recovery of native vegetation.

Where deeper soil layers are susceptible to movement through geological instability or deep zones of saturation, revegetation may offer little prospect for correction. Some lands are not suitable for manipulation or use and we would do well to leave such highly vulnerable landscapes alone. Where roads must cross such areas as slumps, bridges may offer the potential for some use of these lands, depending on the scale of the problem. Often, culverts are very short-term solutions and can cause more problems than they solve. The advice of a local geologist is an essential complement to engineering and soil bioengineering.

THE ROLE OF MAINTENANCE

As important as the revegetation construction methods themselves is the maintenance regime following construction. In the former National Rivers Authority Welsh Region, UK, maintenance staff have demonstrated for over 25 years that cost-effective, ecologically sensitive flood defence can be achieved almost entirely by the vegetation maintenance regime (R. Vivash, 1996, pers. comm.). The long-term success of the "soft-engineering option" ultimately depends on the skill, understanding and dedication of the maintenance staff. Much knowledge already resides in the minds and hands of those who have carried out this work without recognition. River planners, engineers and ecologists will gain enormously from the insights of these people, who can carry forward the multi-function strategy to successful realisation. Sustainable economics, based on longer time intervals like 25-50 years, will point us toward new job creation through creative management of floodplain corridors, developing new industries using sustainably the biological resources and carrying capacity of the floodplain. Here maintenance and resource extraction could become the same activity with multiple consequences.

PRIORITISATION OF RESOURCES

The traditional approach of investment in single functions (e.g. flood defence, water resources, highways, etc.) has tended to give first priority to supplying demand (treating the symptoms), rather than demand management (treating the causes). This has resulted in massive environmental impact, often poorly mitigated, from major civil engineering works. The first priority in single-function projects such as a highway and even multi-purpose projects such as big dams for navigation, irrigation, hydro-electric power and/or flood defence, is given to the huge technical issues. Caring for the local or the upstream and downstream environment has received far lower priority.

With the single-function approach, what priority has been attached to the prevention of problems? The alternative approach is to consider a natural unit with regard to all development functions. This unit is clearly the river catchment, as recognised at the Earth Summit at Rio in 1992 and expressed forcibly in Agenda 21. With an approach to decision-making that has been called the Sustainable Cycle (Gardiner, 1996) integrated catchment management planning becomes a credible vehicle for stakeholder participation and true prioritisation of investment in the community's interests. Then each function becomes linked with other single functions to calculate cumulative effects.

The interests of the community are divided among separate sectors (e.g. water, power, transport), public or private, who rarely communicate with each other. Local barriers to communication and the absence of a holistic preventative approach have accompanied the division of these sectors into individual functions. For example, management of the "water" sector can be subdivided into departments for water quality, flood defence, water resources, water supply, sewerage, surface water management, etc., with separate and non-transferable budgets. Inevitably, each of these departments are concerned to promote their version of how the world should look and can be quite fiercely independent of each other.

Not only will each functional department have its own budget, jealously guarded to maintain the financial capability to fund big projects, but they will rarely admit to having spare resources for "inter-departmental" communication (such as is required for catchment planning, or any other multi-functional initiative) unless this is specified in their remit. In the absence of such specified, multi-functional activities, the interests of the community are actually prioritised in relation to the single function budget demands within individual sectors. Small wonder that professionals in many countries have lost credibility with the communities they sought to serve.

How can this situation be reversed in the interests of sustainable development, if not the recovery of professional standing? Multi-functional activities need a sponsor and they also need specification; they rarely happen spontaneously. One route is to specify multi-functional activities which demand cross-functional communication. Typically a local authority or agency is needed to initiate and then lead the community in increasing dialogue with all possible participants. Such a person or team must be funded to carry out the process of mediating disputes among groups who do not necessarily share values (such as engineers and geomorphologists, for example). For river buffer zones, even "illegitimate" stakeholders such as the local motorcycle gang may need to be heard if they consider some floodplain their recreational "turf", for without their support a good plan may be vandalised and ecological benefits lost. Training in the skills of "Alternate Dispute Resolution" may become indispensable to the ecological engineer of the future.

ENVIRONMENTAL ASSESSMENT AS PROCESS AND DIALOGUE

Another route is to specify the process of decision-making for multi-functional and stakeholder participation; thus, decision-making in projects can use the environmental assessment (EA) process as a means of progressing the dialogue with stakeholders. Plans and programmes, if not policies, can be evolved in a similar way using strategic EA, as mentioned above. Could focusing on the multi-sectoral, multi-functional areas of land care, such as buffer zones, coupled with an appropriate process for decision-making, help the real picture of sustainable development to emerge?

Soil bioengineering in buffer zones can be applied to erosion control, source control and remediation of contaminated or damaged land and rivers (Schiechtl, 1980). In many countries, it is these areas which are rapidly assuming first priority, as realisation dawns that natural resources are being despoiled without thought for the needs of future generations. But few countries can claim to have an institutional structure which facilitates such a multi-sectoral approach. A possible exception is New Zealand, where care for land, water and soil is now integrated into regional authorities. How quickly can our institutional arrangements change to meet the new challenge? How can public and private interests best proceed until the institutions respond?

LAND OWNERSHIP AND WILLINGNESS TO FUND LAND CARE

Private land ownership can be supportive to the environment in the hands of those who understand enough to care for the long-term interests of the land over which they preside. Public ownership can, of course, provide a more sustainable approach, particularly where decision-making is vested in organisations such as the UK's National Trust, or in local authorities which take seriously their lead role in delivery of Agenda 21 (Keating, 1993). In other cases, however, the "tragedy of the commons" often conspires against the more obvious benefits of public ownership. A lack of available funds and/or interest in either control of diffuse pollution or biodiversity exacerbates the problem.

As an example of how private and public interests can combine, the Australian programme of federally-supported "Landcare" represents one of the most successful initiatives on a national scale. Land owners concerned for the state of their land get together to seek a communal approach to the problems of erosion, salinisation, eutrophication etc. They can put up a proposal for remedial action which may attract some federal support (over AUS$ 200 million for 1994/5). The nature of their problems means that these groups tend to be formed by catchment and are engaged in a form of catchment planning. The creation of buffer zones feature strongly in the remedial actions taken by these groups, which have multiplied rapidly over the past 15 years or so.

Another example of a similar programme on a regional basis, which has been running almost as long in California, is the "Urban Stream Restoration Program" run by the California Department of Water Resources in Sacramento. Again, riparian owners can get start-up or enabling funds for communal projects aimed at stream rehabilitation, if they cooperate with local and state agencies.

STRATEGIC "BEST VALUE FOR MONEY"

The fact that the Australian "Landcare" initiative has proven so successful illustrates the general truth that coordinated activity, even when funded only to a limited extent, can prove much more effective and efficient than actions taken by a single landowner in isolation. Usually, this coordinated approach can have strategic benefits for neighbours, usually those living downstream. Therefore, all the beneficiaries are encouraged to support the action, financially or in kind, through the organisation provided by the Landcare initiative.

Such demonstrable "value for money" provides an economic key to the new paradigm of sustainable development. Single-function activities have trouble finding economic support without invoking contingent valuation methods (or public willingness to pay for "goods" which are otherwise intangible in economic terms). Experience of real community participation in local activities, supported by private and public organisations, has taught us what an immense difference there can be between this approach and that of the authoritarian *fait accompli.*

Sadly, single-function projects will remain with us for some time to come. However, enhanced environmental and social values can still be achieved by community participation in these projects. Other benefits to project planning, such as funded monitoring programmes, can improve our data archives and knowledge as well as reducing short- and long-term costs such as planning enquiries and maintenance requirements, respectively.

It is not only the intangible nature of environmental and social welfare which encourages poor decision-making based on tangible elements alone. The longer-term economic analysis, which would allow the longer-term view to be counted, is missing from most if not all current benefit/cost analyses which typically might take only a 30-year term of discounted benefits and costs, associated with a discount rate of 6%, as in the UK. Many social and environmental costs and disbenefits are thus visited on future generations, in direct opposition to the principle of inter-generational equity articulated by the Brundtland Report on sustainable development (Pearce, 1995).

If we are serious about stemming environmental decline, there are many tools available to us to make needed changes. The authors see the trends ahead to be moving toward smaller scale projects, toward increasing efforts to work with natural processes and toward increased communication across all sectors of society. There will be many opportunities in the years ahead to choose between large, hard-engineered projects which treat the symptoms of a catchment malaise and relatively small-scale works that seek to treat the causes. From many sources, we now have enough evidence to know that prevention, rather than cure, is nearly always the best and most economic approach.

CONCLUSIONS

There are several conclusions which arise from this approach:

1. To be “sustainable”, new development must conserve and enhance those natural resource attributes of a catchment that made development initially possible or desirable. In every catchment, there is nearly always massive scope for enhancement of river corridors.

2. The potential for enhancement can be realised by cooperative efforts in the community, especially among local and regulatory authorities, infrastructure management and maintenance organisations and the development industry.

3. GIS can be used to assess the catchment’s potential for environmental enhancement. Comparison of historic air photo images allows assessment of geomorphological change. Ground surveys, including topography, river habitat, landscape and geomorphology, are vital to provide further detail to aerial survey when change is being considered. The use of air photo data can best be integrated with the existing land surface with a GIS database and mapping strategy, providing a useful management tool.

4. Vegetated, created wetlands can provide multiple benefits such as flood water storage, wildlife habitat, fibre extraction, open space for passive recreation and other amenities. These multiple functions provide a broader range of benefits, increasing the benefit-cost ratio and making wetland treatment of water more economically feasible than a single purpose approach. Much the same reasoning lies behind the use of vegetation in source control, which is the control of runoff at or near the point of rainfall. Such strategies offer opportunities for creative soil bioengineering design, using material ranging from porous pavements, turf and grass grids, natural fibre geotextiles, from the grasslike sedges and rushes, to willows and other wet-tolerant woody shrubs and trees.

5. Regional assessment of upland problems, based on local geology and the stream network, can be used to identify those areas which have the greatest erosive potentials; steep slopes and the margins of existing upland channels. The greatest challenge in upland erosion control is correct identification of the problem.

6. The option of allowing vegetation to recover by removal of the problem, such as cattle overgrazing, should be given first consideration, after which the option of “allowed natural recovery” should always be considered.

   Bringing vegetation into river management first implies a willingness to give a little more room for Nature. Working with inherent river processes is less costly than trying to command the flood. The direction of intelligent river management is now toward integration of the river channel with its floodplain.

8. The long-term success of the "soft-engineering option" depends on the skill, understanding and dedication of the maintenance staff. Planning, design and construction should include the maintenance staff as full partners on the team.
9. Land use planning and economic incentives are among the first tools of ecological engineering, so that we do not spend precious resources protecting flood-prone lands.
10. Sustainable economics, based on longer time intervals like 25-100 years, will point us toward new job creation through creative management of floodplain corridors, developing new industries sustainably using biological resources of the floodplain.
11. The key to successful decision-making at any scale is to involve the stakeholders to the extent that ownership of the *process,* as well as the resulting activities, is shared.
12. Decision-making in projects can use the environmental assessment (EA) process as a means of progressing the dialogue with stakeholders. Plans and programmes, if not policies, can be evolved in a similar way using strategic EA.
13. With an approach to decision-making called the Sustainable Cycle, integrated catchment management planning becomes a credible vehicle for stakeholder participation and true prioritisation of investment in the community's interests.
14. Experience of real community participation in local activities, supported by private and public organisations, has taught us what an immense difference there can be between this approach and that of the authoritarian *fait accompli.* Such demonstrable "value for money" provides an economic key to the new paradigm of sustainable development.

REFERENCES

Hall, M.J., Hockin, D.L. and Ellis, J.B. (1993) *Design of flood storage reservoirs.* Construction Industry Research and Information Association (CIRIA) jointly with Butterworth-Heinemann Ltd., Oxford

Coppin, N.J. and Richards, I.G. (1990) *Use of vegetation in civil engineering.* Butterworths, Construction Industry and Research Information Association, London.

DWVK (1996) *Fluss und Landschaft – Okologische Entwicklungskonzepte.* Deutscher Verband fur Wasserwirtschaft und Kulturbau e.V., Bonn.

Gardiner, J.L. (1988) Environmentally sound engineering; examples from the Thames catchment. In Petts, G.E. and Wood, R. (Eds.) *Regulated Rivers,* **2**(3), 445-471, John Wiley & Sons, Chichester.

Gardiner, J.L. (1990) River catchment planning for land drainage and the environment. *Journal of the Institution of Water and Environmental Management,* **4**(5), London.

Gardiner, J.L. (1996) The Use of EIA in delivering sustainable development through integrated water management. *European Water Pollution Control,* **6**(1), pp. 50-59.

Gray, D.H. and Leiser, A.T. (1982) *Biotechnical slope protection and erosion control.* Van Nostrand Reinhold Co. New York.

Gray, D.H. and Sotir, R. (1992) Biotechnical stabilisation of a highway cut slope. *ASCE J.Geotech. Div.,* pp. 1395-1409.

Gregory, K.J. and Walling, D.E. (1973) *Drainage Basin Form and Process.* Edward Arnold, London.

Keating, M. (1993) *The Earth Summit's Agenda for Change: a plain language version of Agenda 21 and the other Rio Agreements.* Centre for Our Common Future, Geneva.

Kinsley, M.J. and Lovins, L.H. (1995) *Paying for growth, prospering from development.* Rocky Mountain Institute, Old Snowmass, Colorado.

Lowrance, R., Altier, L.S., Newbold, J.D., Schnabel, R.R., Groffman, P.M., Denver, J.M., Correll, D.L., Gilliam, J.W., Robinson, J.L., Brinsfield, R.B., Staver, K.W., Lucus, K. and Todd, A.H. (1995) *Water Quality Functions of Riparian Forest Buffer Systems in the Chesapeake Bay Watershed.* U.S.E.P.A. Chesapeake Bay Program. Technology Transfer Report. Annapolis, MD.

NRATR (1995) *Thames 21: a planning perspective and a sustainable strategy for the Thames Region.* National Rivers Authority, Thames Region, Reading.

Newson, M.D. (1992) *Land, Water and Development.* Routledge, London

Pearce, D. (1995) *Blueprint 4.* Earthscan Publications, London

Perala, N.C (in press) *Protocols for Soil Bioengineering; Site Assessment and Monitoring for Streambank and Floodplain Revegetation.* Presented at River Restoration '96 conference, Silkeborg, Denmark.

RSPB, NRA and RSNC (1994) *The New Rivers and Wildlife Handbook.* The Royal Society for the Protection of Birds, Sandy, Bedfordshire.
Schiechtl, H. (1980) *Bioengineering for land reclamation and conservation.* Univ. Alberta Press.
Therivel, R., Wilson, E., Thompson, S., Heaney, D. and Pritchard, D. (1992) *Strategic Environmental Assessment.* Earthscan, London.
USDA (1992) Soil bioengineering for upland slope protection and erosion reduction. *Engineering Field Handbook,* Part 650. U.S. Dept of Agriculture, Washington.

Note

Prichard, D. (1996) Pers. Comm. Range Ecologist for the USDI Bureau of Land Management, Denver, Colorado, USA. See the Interagency Stream Restoration Handbook, 1996, from BLM.
Roseboom, D. (1996) Pers. Comm. Engineer with Illinois State Water Survey, Peoria, IL.

Encouraging Implementation of Riparian Buffer Schemes – The New Zealand Experience

29

A.B. Cooper[1], N.I. Ngapo[2], T.G. Parminter[3] and M.J. Stroud[1]

[1]*National Institute of Water and Atmospheric Research, Hamilton, New Zealand*
[2]*Environment Bay of Plenty, Whakatane, New Zealand*
[3]*AgResearch, Hamilton, New Zealand*

Abstract

Encouraging widespread implementation of riparian buffer schemes requires that legal, political, technical, economic and social constraints are overcome. New Zealand's Resource Management Act (1991) has as its purpose "to promote the sustainable management of natural and physical resources" while "avoiding, remedying or mitigating any adverse effects on the environment". This legislation, along with subsequent Government policy that has been consistent with this Act, has promoted a strong interest in riparian buffers from both resource management agencies required to implement the Act (Regional Councils) and resource users seeking to meet their legal obligations under the Act. Scientific research in New Zealand (and elsewhere) has shown the benefits of riparian buffers sufficiently to support their promotion as a management tool and guidelines to assist in their use have been published. Regional Councils are now encouraging implementation by environmental education and community consultation in preference to rules-based or compensatory approaches. Community "ownership" of riparian buffer schemes is seen as vital to their widespread adoption and on-going maintenance. New Zealand farmers rank riparian buffers highly in terms of meeting their resource management goals but negatively in terms of meeting their profitability and lifestyle goals. Future research will need to consider innovative ways in which riparian buffers can be designed to reduce this conflict.

INTRODUCTION

"....I have become satisfied that the destruction of trees bordering on these streams and the changed condition of the banks produced thereby, has resulted in the destruction of the natural harbours or hiding places of the trout....." (Van Cleef, 1885).

Bush Stream

"Tall trees covering the sun
Cold and scary
Ferns growing
Bumpy logs
Skeleton leaves
A home for fish
Awhi our bush."

Megan Cooper (Aged 6)
(Awhi – Maori for preserve)

The influence of riparian zones on streams has been known for some considerable time and is intuitively logical to even the youngest of us. The logic of protecting, restoring or managing riparian zones to mitigate the in-stream effects of wider land use activities is well-supported scientifically. In addition, there is recognition that riparian zones can provide an important habitat for terrestrial wildlife and act as a corridor for the passage of species between suitable habitat patches (e.g.,

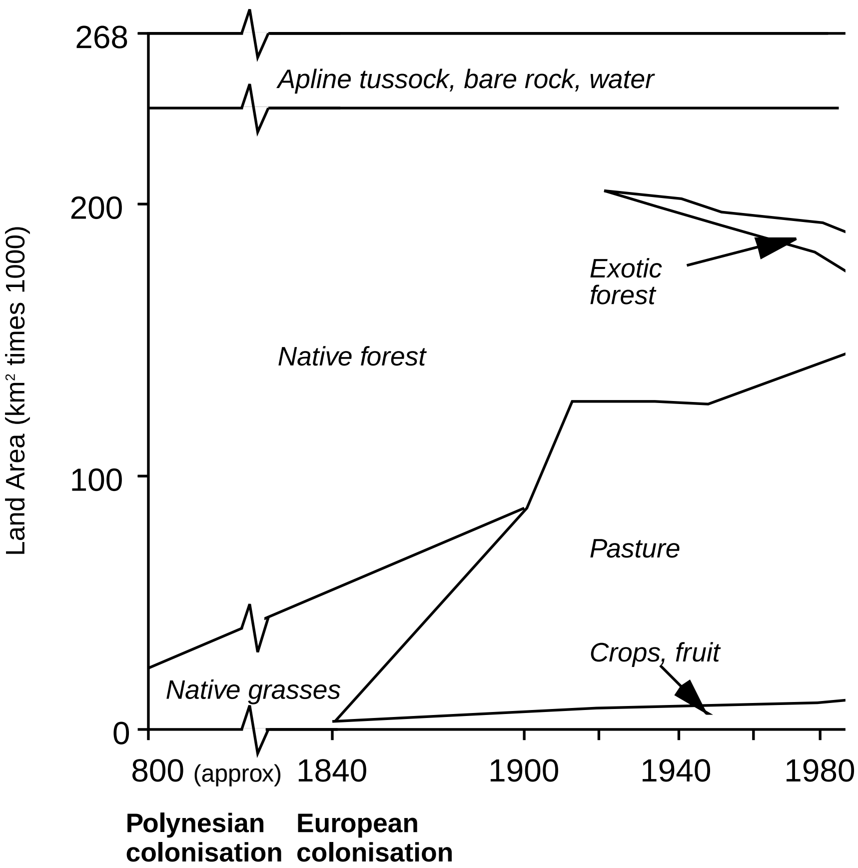
Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0

indigenous forest) in an otherwise unsuitable landscape. If these ecological perspectives were the only consideration, then all our developed landscapes would be traversed by contiguous buffer strips of riparian vegetation. That this is not so, merely demonstrates the obvious; that there are socio-economic constraints that limit implementation of ecologically-sound and scientifically-validated land management practices (Novotny, 1988). In this paper we present the New Zealand experience on encouraging implementation of riparian buffer schemes and discuss the constraints that need to be overcome for such implementation to become common place.

BACKGROUND

Development of New Zealand's land for productive use is recent when compared to most other countries (Fig. 1). Furthermore, for most New Zealand aquatic ecosystems (excepting, perhaps, lowland streams) there are sufficient remnants of indigenous land cover to enable the impacts of catchment development on receiving waters to be unambiguously determined. The majority of land use change has been from indigenous podocarp-hardwood forest or tussock to pastoral agriculture which now occupies about 50% of New Zealand's land area and supports 49 million sheep, 5 million beef cattle, 4 million dairy cows and 1 million deer. These livestock produce an effluent loading equivalent to 150 million people, which places their environmental significance into perspective, given New Zealand's human population of approximately 3.4 million.

Figure 1. History of land use changes in New Zealand.



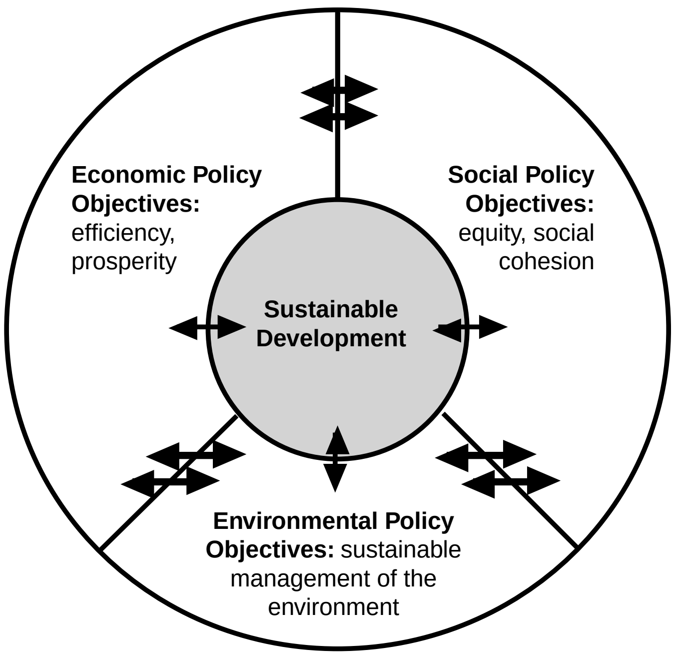
The receiving water impacts of this land use change are significant, although they appear to be frequently less severe than that observed elsewhere. For example, New Zealand's surface waters have a median nitrate concentration of 0.105 g m^{-3}, and rarely exceed 1 g m^{-3} (Smith and Maasdam, 1994), well below similar statistics for other agriculturally-developed countries (Meybeck *et al.*, 1989). The impacts of catchment development on New Zealand's streams, lakes and estuaries have been well-documented in various reviews of research and reports to resource management agencies and Government departments. Of particular note is that the link between catchment development and degraded receiving waters was reported on at least as early as the 1960s (Fish, 1969), and evidence continued to be added through the 1970s (e.g., McColl *et al.,* 1975; White, 1977) and 1980s (e.g., McColl and Hughes, 1981; Hume and McGlone, 1986; Various, 1986). Despite this weight of evidence, there was not widespread acceptance amongst New Zealanders that a problem existed and commitment to remedial action was patchy and sporadic.

The 1990s has seen a major change within New Zealand, initiated by enactment of the Resource Management Act (RMA) in 1991, which has as its purpose "to promote the sustainable management of natural and physical resources" while "avoiding, remedying or mitigating any adverse effects on the environment". The RMA has brought together in one statute the management of land and water, thereby providing legislative support for considering off-site impacts of land use and the mechanisms for dealing with such impacts. This legislation, along with subsequent Government policy that has been consistent with this Act, has promoted a strong interest in riparian buffers from both resource management agencies required to implement the Act (Regional Councils) and resource users seeking to meet their legal obligations under the Act.

POLICY CONSTRAINTS

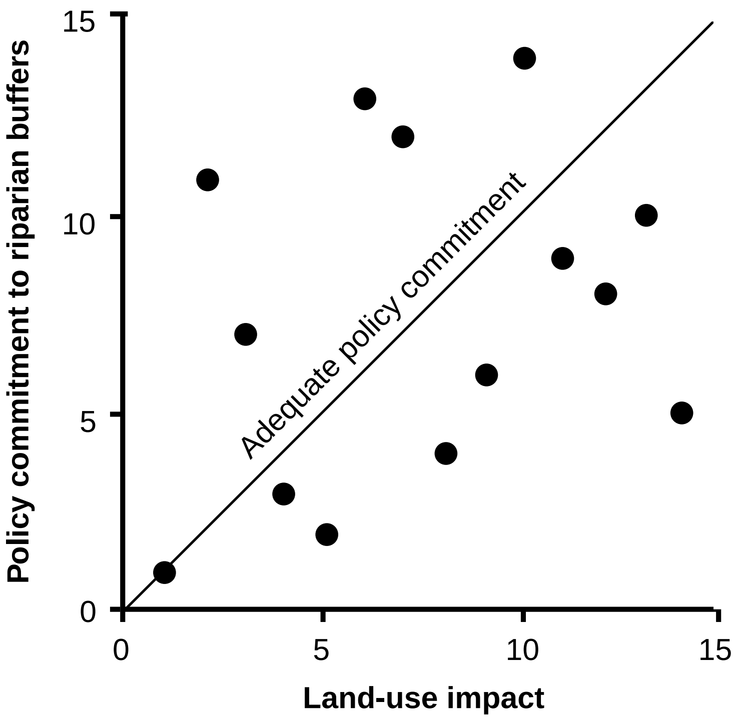
During the 1990s, New Zealand's policies have consistently placed environmental goals alongside economic and social goals, with clear statements that economic growth and social development must occur within a framework of sustainable resource management (Fig. 2). This linking of socio-economic goals with environmental goals has been important, with government agencies traditionally responsible for promoting increased production (e.g., Ministries of Agriculture and of Forestry, Producer Boards) now explicitly required to take into consideration the environmental effects of policies they may promote to the sectors they serve. The changes resulting from this requirement are having far-reaching effects, with government agricultural agencies and large agri-business commissioning studies to define their impacts on the environment and how these may be reduced (e.g., Smith *et al.,* 1993).

Figure 2. New Zealand's integrated policy framework for sustainable development (from MfE 1995).



The RMA devolves considerable responsibility for implementation of its provisions to regional government (Regional Councils). Regional Councils have a mandatory requirement to prepare regional policy statements, and to prepare regional plans where there are significant conflicts between the use of land or water (present or proposed) and the protection, restoration or enhancement of resources. The Act's principle of "avoiding, remedying or mitigating any adverse effects on the environment" suggests that regional councils with significant land use impacts on receiving waters need to promote the use of riparian buffer zones as one tool for achieving their policy goals. We have conducted a desk-top analysis, admittedly subjective, on the relationship between the relative degree to which riparian buffers are promoted in a region's policies and plans and the relative degree to which each region's waters are impacted by land use. Information for the latter was derived from a variety of sources, including the region's own monitoring programmes, the national river water quality monitoring network (Smith and Maasdam, 1994) and the summary of information provided in Smith *et al.* (1993). This analysis shows that relative policy commitment to the use of riparian buffers does not always match with the relative size of the land use impact (Fig. 3). While a low policy emphasis on riparian zone management may be appropriate in those regions where the impacts of land use are relatively minor, a lack of promotion where impacts are high suggests that these regions may find it difficult to fulfil their obligations under the RMA.

Figure 3. Relationship between regional council policy commitment to the use of riparian buffer zones and the relative impact of land use on receiving waters.



While policy commitment is important, it is the achievement of this policy commitment by on-the-ground action that is the ultimate measure of success. It is too soon to judge success in these terms as the regional councils are currently at various stages in implementing their policies and plans. Some regional councils have a long history of promoting and implementing riparian buffer schemes and these councils have continued this commitment within the new framework of the RMA. For example, in 1993 Environment Bay of Plenty adopted a policy which committed it to a 30-35 year programme of riparian protection works, including in excess of 1650 km of fencing. In the 1994/95 financial year, 37 km was undertaken at a cost of $257,000NZ, approximately $7,000 per kilometre. This expenditure provided for fencing materials and riparian trees, but not compensation for loss of productive land.

Although the RMA provides regional councils with a policy framework for implementing riparian buffer schemes, Section 32 of the Act also places a requirement on councils to justify their use, consider any alternatives to achieving the same objectives, and the relative costs and benefits of these alternatives. Meeting this onus requires that councils have scientific research that supports their proposals and this need was a major factor in the development of interim guidelines for riparian management (Collier *et al.*, 1995). How well current scientific knowledge meets the needs of resource managers seeking to implement riparian buffer schemes is discussed in "Technical Constraints".

SOCIAL CONSTRAINTS

> *"I have a right to do what I want on my land, and if I am to be constrained by the community, I have a right to be compensated."* Anon., Farmer

One argument put forward by farmers is that if the community desires an improvement in water quality then the community should compensate the farmer for the costs associated with modifying land use practices. However, New Zealand property rights law is clear, with owners having the right to use, enjoy and take profits from the land, but they do not have the right to do with the land anything they wish. This means that restrictions on land use activities brought in as a rule in a regional plan under the RMA are legally enforceable. It seems likely that as more plans become operative under the RMA, more restrictions will be placed on the way New Zealanders use their land. However, whilst the RMA provides such regulatory powers, there is now increasing acceptance in New Zealand that emphasis on a rules-based approach will not meet the sustainability goals of the RMA and that emphasis on encouraging voluntary adoption of environmentally-sound land management practices is the way forward (O'Brien, 1994; MfE, 1996). While regulatory controls have proved their value in controlling point source pollution, they are a cumbersome tool for controlling non-point source pollution, being difficult to devise and expensive to monitor and enforce.

Farming leaders have generally been supportive of the purpose of the RMA, viz. sustainable management of resources, but critical of the way some regional councils initially approached implementing the Act which was seen as over-regulatory, non-consultative and ignorant of farming operations (Simeonidis, 1994). New Zealand farming leaders have made the generalisation that farmers are independent by nature, resent rules and resent bureaucratic intervention but are willing to accept responsibility for their own actions and have a desire to be good stewards of the land for future generations (e.g., Pinnell, 1994). Farmers have sought to participate in the process of developing strategies for reducing the off-site effects of farming and this participative approach is now becoming the norm. During the process it has become apparent that farmers prefer a voluntary approach, supported by rules to bring the wayward farmers into line so as not to compromise the efforts of others (MfE, 1996). There is now increasing recognition that this voluntary approach, when supported by education initiatives, making expert help available, and limited incentives (e.g., rates relief, provision of riparian plants), is the only approach to achieving large-scale implementation of riparian buffer zone management that is effective in the long-term. Farming leaders recognise the need to be proactive in developing environmentally-sound land use practices so that the solutions arrived at are practical at the farm level. A key tool in encouraging implementation of better farming practices is the formation of farmer-initiated landcare groups, where information is shared amongst farmers and where scientists and resource managers are invited to attend to provide their expertise.

In encouraging voluntary adoption, extension strategies can be used to provide direction and motivation for landowners to establish riparian buffers. Although education initiatives may result in farmers being more aware of the need to change farming practices to improve resource condition, they may still be slow to implement change due to a number of constraints (Table 1). Of these constraints, a lack of resources, or planning to overcome apparent lacks, feature prominently. To encourage changed behaviour in landowners, where they are fully informed as to what is desirable behaviour, requires knowledge of the incentives and disincentives around their current practices. Farmers have a number of personal goals and aspirations for themselves and their properties (Parminter *et al.*, 1996).

Commonly these include maintaining farming profitability and farming lifestyles, as well as developing business viability and natural resource maintenance. The resolution of these goals in farming decisions requires some compromise between them as implementing any goal area in isolation will create conflicts with the others (Table 2). It is the perception of conflicts between resource management goals and goals for profitability and farming lifestyles that currently limits the voluntary adoption of riparian buffers by farmers. Some farmers' experiences of riparian buffers are that they are costly, remove good land from production, and require extra time to maintain them (Meister *et al.*, 1992). To encourage adoption, these constraints do not have to be completely removed, but they do need to be overcome sufficiently for farmers to perceive that they can obtain a net benefit (O'Brien, 1994; Parminter *et al.*, 1996).

Table 1. Results of a survey of farmers' perceptions of the key constraints to land conservation (multiple answers allowed).

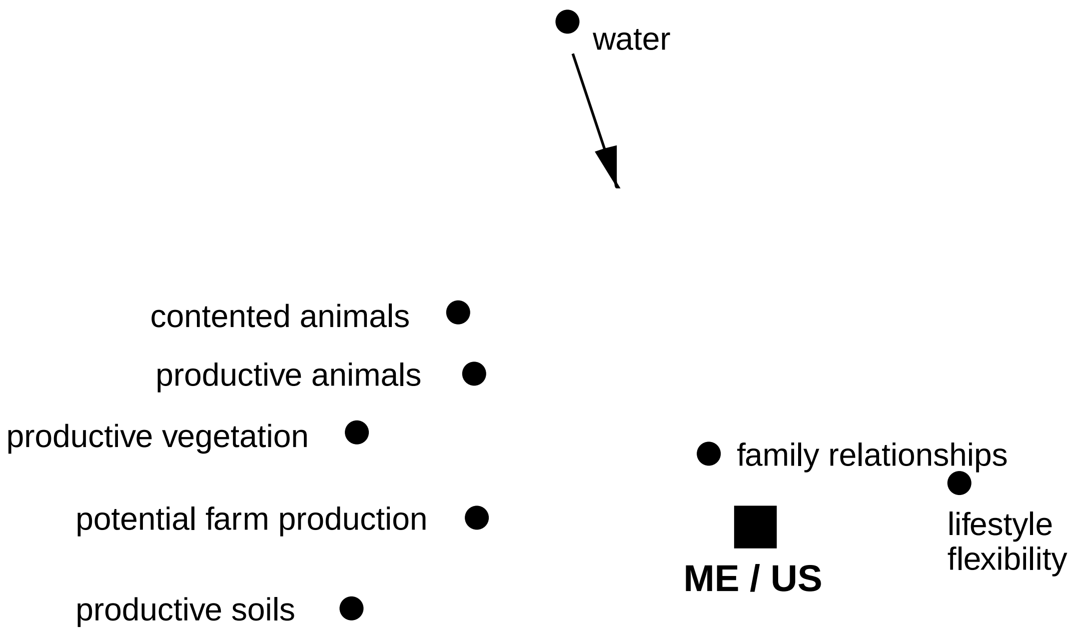
Constraint	*Frequency of Response %*
Lack of money	95
Lack of time	26
Attitudes	21
Knowledge	10
Season/climate/weather	6
Other	21
Don't know	2
None	1

Table 2. Farmers' priority rankings for management objectives that will achieve separate farming goals of resource management, annual profitability and satisfying lifestyle (a selection from 17 objectives, see Parminter *et al.*, 1996).

Management Objectives	*Farming Goals*		
	Resource Management	*Annual Profitability*	*Satisfying Lifestyle*
Stock health and condition	10	2	1
Targets for stock performance	16	1	3
Protection of streambanks and waterways	1	17	15

Research in New Zealand amongst landcare group members has shown that, initially, farmers' water quality objectives were isolated from their other farming objectives (Fig. 4). Figure 4 is a two-dimensional belief and attitude map describing the associations between various farming goals (Walfel and Fink, 1980). In the figure those objectives that are mutually supportive (such as "farming profit" and "animal productivity") are close together. For these particular farmers, whilst they had an objective to maintain and improve water quality that objective would have been easily left unfulfilled because it hadn't been integrated into their daily decision making. In the 12 months that the landcare group (comprising both farmers and scientists) has been operating, these farmers have begun to understand how much water quality is affected by how they manage other natural resources on the property such as soils, vegetation and livestock. As a result, the effect of their management upon water quality is becoming much more integrated into their daily decision making, changing their attitude towards water quality in the direction of the arrow in Fig. 4. Their more positive attitude means that they are much more likely to implement strategies for protecting water, including the establishment of riparian buffers.

Figure 4. An attitude and belief map of the relationships between farming objectives for a dairy farming partnership. Proximity of objectives to one another shows the farmers' belief of how closely related these objectives are whilst their closeness to the Me/Us square shows the farmers' attitude as to how important each objective is to them.



TECHNICAL CONSTRAINTS

The international literature contains many studies that have shown the environmental benefits of riparian buffer zones. In general, the considerable New Zealand research has supported this conclusion (Quinn *et al.*, 1993). This research has been extremely valuable in scientifically justifying the principle of riparian buffers. However, it is our contention that future research must turn to providing information to those already convinced of the principle but who now require detailed methodologies for implementation. In our experience the questions most frequently asked by would-be implementers of riparian buffer schemes are deceptively simple: where should I put them? how wide should they be? what should I plant in them? how much will it cost? how do I look after them? and what will the benefits (and negative impacts) be and how long will it take for them to occur? Many of these questions require a predictive approach, where prior research is integrated and findings are transferred to new situations. There is uncertainty with these predictions but uncertainty is common in environmental decision-making and, perhaps, cannot be posed as an excuse for inactivity.

Scientifically, riparian zones pose many interesting challenges to our current understandings of how boundaries between ecosystems function and how this functioning can be altered by various natural and anthropogenic disturbances. While there is much to be gained by long-term curiosity-driven research into the structure, function and dynamics of the land-water interface, the urgent needs of resource management has seen New Zealand studies on riparian zones being issues-driven. Experimental studies in New Zealand (and we would suggest elsewhere) have typically been short-term, small in scale (plot to small catchment), compared inputs with outputs, and addressed one of the issues associated with riparian buffer zones in isolation from others. Observational research in New Zealand, which has compared stream condition with and without riparian buffers (Quinn *et al.*, 1992; Smith, 1992; Howard-Williams and Pickmere, 1994; Williamson *et al.*, 1996), have been either spatial or temporal comparisons, but not both. There are difficulties in space-for-time substitution studies (Pickett, 1989) and in transferring site-specific time series information to other locations (Naiman *et al.*, 1995; p.85). There is a need for studies on riparian buffers that combine both temporal and spatial response within an appropriate statistical design (Underwood, 1991 and 1993). It is unfortunate that past opportunities for this type of approach have been missed in New Zealand, where the riparian buffer schemes that have been implemented would have provided the large-scale experiment for a learning-by-doing approach (Walters and Holling, 1990).

While New Zealand studies have generally shown that riparian buffers have a positive effect on streams, there is a need to exercise some caution as several studies have either shown that negative effects can occur (Smith, 1992) or that raise questions about the long-term sustainability of buffering processes (Cooper *et al.*, 1995). There is a need for research that examines the issue of sustainability of contaminant removal processes within the riparian zone. We hypothesise that for conservative materials (e.g., phosphorus, heavy metals), available “storage” within the riparian zone will become saturated and with continued inputs from upslope there will be an inevitable transport to receiving waters. By comparison, for non-conservative materials (e.g., bacteria, degradable pesticides) we hypothesise that the extra residence time afforded by entrapment within the riparian zone will allow permanent removal mechanisms to operate and buffer the streams indefinitely providing input rates are matched to removal rates.

Resource managers in New Zealand have identified their top priority as the need for research to be accessible and easily understood by those implementing riparian buffer schemes (Smith, 1993) and in response to this need we have produced an interim set of riparian guidelines (Collier *et al.*, 1995). In preparing these guidelines it became obvious to us that there is a need for research to provide tools for the optimal design of riparian buffers to achieve the range of water quality and ecological objectives that are typically put forward. On the same theme, there are insufficient tools available on how to assess the relative costs and benefits of riparian management versus other changes in land management (e.g., changes in fertiliser practices) that have the potential to achieve the same objectives. Both of these research gaps will require a combination of scientific skills, including predictive modelling, aquatic ecology, agronomy and economics.

We consider that there is a need for land users and resource managers to be involved in the research process, from the design phase, through execution, to the write-up of findings. There is also a need for researchers to be involved in the implementation process. There are at least two reasons for this. Firstly, the application of research findings by implementers (primarily farmers and staff in resource management agencies) will be most rapid and effective when implementers are involved and can take ‘ownership’ of the results. Secondly, the involvement of researchers in the implementation process will provide them with a greater appreciation of the issues faced by farmers and help in developing research hypotheses for testing that are most relevant to end-user needs.

CONCLUSIONS

New Zealand’s environmental legislation and policies provide a suitable framework for resource management agencies to promote the use of riparian buffers to help achieve their sustainability goals. It is also apparent that the New Zealand farming community does not need to be further convinced of the potential off-site impacts of their activities and the importance of carefully managing streamside areas. There has also been considerable research in New Zealand that provides scientific demonstration of the various environmental benefits of riparian buffers. These positives provide a sound platform from which to encourage voluntary adoption of riparian buffers. Key constraints that remain include: providing practical means by which farmers can manage riparian buffers to satisfy sufficiently both their profitability and resource management goals; providing decision-support tools for resource management agencies to decide on the most cost-effective design for a riparian buffer scheme and whether riparian buffers are the best alternative for meeting a particular environmental goal; and providing information on the ability of riparian buffers to sustain their desired functions in the long-term. To overcome these constraints the research required needs to be conducted in a partnership between scientists and implementers.

REFERENCES

Collier, K.J., Cooper, A.B., Davies-Colley, R.J., Rutherford, J.C., Smith, C.M. and Williamson, R.B. (1995) *Managing riparian zones: A contribution to protecting New Zealand's rivers and streams* (Two volumes). Department of Conservation, Wellington, New Zealand.

Cooper, A.B., Smith, C.M. and Smith, M.J. (1995) Effects of riparian set-aside on soil characteristics in an agricultural landscape: Implications for nutrient transport and retention. *Agriculture, Ecosystems and Environment,* **55**, 61-67.

Fish, G.R. (1969) Lakes: the value of recent research to measure eutrophication and to indicate possible causes. *Journal of Hydrology* (NZ), **8**, 77-85.

Howard-Williams, C. and Pickmere, S. (1994) Long-term vegetation and water quality changes associated with the restoration of a pasture stream. In: Collier, K.J. (Ed.) *Restoration of Aquatic Habitats.* Department of Conservation, Wellington, New Zealand, 93-109.

Hume, T.M. and McGlone, M.S. (1986) Sedimentation patterns and catchment use change recorded in the sediments of a shallow tidal creek, Lucas Creek, Upper Waitemata Harbour, New Zealand. *New Zealand Journal of Marine and Freshwater Research,* **20**, 677-687.

McColl, R.H.S. and Hughes, H.R. (1981) *The effects of land use on water quality – A review.* Water and Soil Miscellaneous Publication 23. 59p. Ministry of Works and Development, Wellington.

McColl, R.H.S., White, E. and Waugh, J.R. (1975) Chemical run-off in catchments converted to agricultural use. *New Zealand Journal of Science,* **18**, 67-84.

Meister, A.D., Weber, J.A. and Lynch, R.J. (1992) *Benefits and Costs of Soil Conservation Works in the Bay of Plenty Region.* A report prepared by Department of Agricultural Economics and Business, Massey University, Palmerston North, for Environment Bay of Plenty, Whakatane, New Zealand. 102p.

Meybeck, M., Chapman, D.V. and Helmer, R. (1989) *Global Freshwater Quality.* Global Environmental Monitoring System, World Health Organization – United Nations Environmental Programme. Blackwells, Oxford. 306p.

MfE (1996) *Sustainable Land Management: A Strategy for New Zealand.* Ministry for the Environment, Wellington, New Zealand.

Naiman, R.J., Magnuson, J.J., McKnight, D.M. and Stanford, J.A. (Eds.) (1995) *The Freshwater Imperative: A Research Agenda.* Island Press, Washington D.C. 165p.

Novotny, V. (Ed.) (1988) *Political, Institutional and Fiscal Alternatives for Nonpoint Pollution Abatement Programs.* Marquette University Press, Milwaukee.

O'Brien, M. (1994) Community perspectives of riparian management and restoration: A case study in Marlborough. In: Collier, K.J. (Ed.) *Restoration of Aquatic Habitats.* Department of Conservation, Wellington, New Zealand, 145-163.

Parminter, T.G., Wedderburn, M.E. and Spijkerbosch, P.J. (1996) The importance of a participatory approach to resolving issues of sustainable agriculture. *Proceedings of the New Zealand Grassland Association,* **57**, 15-19.

Pickett, S.T.A. (1989) Space-for-time substitution as an alternative to long-term studies. In: Likens, G.E. (Ed.) *Long-term Studies in Ecology,* Springer-Verlag, New York, 110-135.

Pinnell, G. (1994) *Land use controls: Reaction of farmers.* Proceedings of the New Zealand Association of Resource Management Annual Conference, Hamilton, New Zealand.

Quinn, J.M., Williamson, R.B., Smith, R.K. and Vickers, M.V. (1992) Effects of riparian grazing and channelisation on streams in Southland, New Zealand. 2. Benthic invertebrates. *New Zealand Journal of Marine and Freshwater Research,* **26**, 259-269.

Quinn, J.M., Cooper, A.B. and Williamson, R.B. (1993) Riparian zones as buffer strips: A New Zealand perspective. In: Bunn, S.E., Pusey, B.J. and Price, P. (Eds.) *Ecology and Management of Riparian Zones in Australia,* LWRRDC Occasional Series No 05/93. LWRRDC, Canberra and CCISR, Griffith University, Marcoola, Queensland, 53-88.

Simeonidis, T. (1994) Effects of implementing the Resource Management Act on the farming industry. In: *Implementing the Resource Management Act,* Proceedings of the Sixth National Water Conference, Hamilton, New Zealand, 25-31.

Smith, C.M. (1992) Riparian afforestation effects on water yields and water quality in pasture catchments. *Journal of Environmental Quality,* **21**, 237-245.

Smith, C.M. (1993) *Perceived riverine problems in New Zealand, impediments to environmentally sound riparian zone management and the information needs of managers.* NIWA Publication 24.

Smith, C.M., Wilcock, R.J., Vant, W.N., Smith, D.J. and Cooper, A.B. (1993) Towards Sustainable Agriculture: *Freshwater Quality in New Zealand and the Influence of Agriculture.* MAF Policy Technical Paper 93/10. Ministry of Agriculture and Fisheries, Wellington, New Zealand.

Smith, D.G. and Maasdam, R. (1994) New Zealand's National River Water Quality Network. I. Design and physico-chemical characterisation. *New Zealand Journal of Marine and Freshwater,* **28**, 19-35.

Underwood, A.J. (1991) Beyond BACI: Experimental designs for detecting human environmental impacts on temporal variations in natural populations. *Australian Journal of Marine and Freshwater Research,* **42**, 569-587.

Underwood, A.J. (1993) The mechanics of spatially replicated sampling programmes to detect environmental impacts in a variable world. *Australian Journal of Ecology,* **18**, 99-116.

Van Cleef, J.F. (1885) How to restore trout streams. *Transactions of the American Fisheries Society*, **14**, 50-55.
Various (1986) Effects of land development on aquatic ecosystems. *New Zealand Agricultural Science*, **20** (2), Special Issue.
Walfel, J. and Fink, E.L. (1980) *The Measurement of Communication Processes: Galileo Theory and Method.* Academic Press, San Francisco.
Walters, C.J. and Holling, C.S. (1990) Large-scale management experiments and learning by doing. *Ecology*, **71**, 2060-2068.
White, E. (1977) *Eutrophication in perspective.* Proceedings of the 16th Technical Conference of the New Zealand Fertiliser Manfacturers' Research Association 1, 74-78.
Williamson, R.B., Smith, C.M. and Cooper, A.B. (1996) Watershed riparian management and its benefit to a eutrophic lake. *Journal of Water Resources Planning and Management (ASCE)*, **122**, 24-32.

Buffer Zones: Current Concerns and Future Directions

30

N.E. Haycock[1], G. Pinay[2], T.P. Burt[3] and K.W.T. Goulding[4]

[1] *Quest Environmental, PO Box 45, Harpenden, Herts., AL5 5JL, UK*
[2] *Laboratoire d'évolution des systéms naturels et modifiés, URA ECO Bio 1853, Campus de Beaulieu, Université de Rennes I, 35042 Rennes Cedex, France*
[3] *Department of Geography, University of Durham, Science Laboratories, South Road, Durham, DH1 3LE, UK*
[4] *IARC Rothamsted Experimental Station, Harpenden, Herts., AL5 2JQ, UK*

Abstract

This concluding chapter draws together a summary of the main discussions and observations at the conference and comments on the future direction of work on this subject. There is general agreement that buffer zones hold enormous potential for the control of water-based pollution and for conservation and catchment biodiversity; they should, as a consequence, be applied more widely. Although there is a reasonably clear understanding of how and why buffer zones control pollution, many issues remain to be resolved and critical scientific information is still needed. Few studies have undertaken complete hydrological and chemical balances or examined more complex hydrogeological environments and some key nutrient processes within buffer zones still need to be resolved. Critical to implementation is the need for a catchment perspective. At the same time, locating risk areas and focusing efforts within the drainage basin is vital. At the farm scale we need better tools to design and assess the full impact of a buffer zone. The few existing procedures focus on how to implement buffer zones for the protection of water and show great potential for the way forward.

INTRODUCTION

The clear aim of this book was to explore the processes that occur in any environment acting as a buffer zone. Having established the potential of different habitats to act as buffer zones, we have sought to explore how a habitat could be protected and managed within the context of current farming regimes and pressures.

There is a desperate need to create and restore a range of habitats within a catchment in order to buffer the impacts of both diffuse and point source pollution. The primary focus of the conference was, however, diffuse pollution, the management of which is complicated. By definition, diffuse, or non-point source pollution is spatially ill-defined and the principal contaminants (the current concerns being nitrogen: N, phosphorus: P, pesticides and sediments) are delivered to the stream environment along various dynamic pathways. Locating buffer zones in an area where there is the potential for diffuse pollution and ensuring that they can cope with the quantity and type of pollution and how it is delivered is a major challenge. The differences between diffuse and point source pollution are obvious, but there are difficulties in modelling an ecosystem's response to both.

With point source pollution, whether from urban storm water drains or waste water effluent, contaminant levels tend to be high and the ecotoxicological impacts of untreated waste profound. Diffuse pollution concentration by comparison may be significantly lower but the cumulative impact, both in total mass and long-term effect, can be greater. The solution for each type may be very different. Buffer zones clearly have a role in the management of diffuse pollution but, as we will emphasise throughout this chapter, the concept will most probably provide one technique within a total strategy for the long-term control of diffuse pollution. In many cases it will be important to combine buffer zones with other tools and techniques (both in-field and off-field).

Buffer Zones: Their Processes and Potential in Water Protection. Edited by N.E. Haycock, T.P. Burt, K.W.T. Goulding and G. Pinay. ISBN 0 9530051 0 0

The processes that occur within buffer zones are varied. In the first part of the book the principal processes accounting for N, P, sediment and pesticide transformations were reviewed and in many of the chapters consistent messages emerge. Many of the key processes are bacterially mediated but the opportunity for these bacterial processes to influence contaminant levels is highly dependent on the hydrology of the buffer zone environments and the source and transportation of contaminants (Burt, this volume).

BIOLOGICAL PROCESSES

The processing of nitrate within riparian environments is discussed in depth within this book, and it is generally concluded "that riparian buffer zones are the most important factor controlling entry of N, particularly nitrate, from agricultural land into surface water in humid regions" (Gilliam *et al.*, this volume). Most authors who have commented on nitrate transformations in riparian zones consider that denitrification of nitrate to di-nitrogen gas is the single most important process. Indeed the processing of nitrate in ponds, wetlands, stream beds, wet meadows and shelter belts is highly dependent on the denitrification process. However, it is apparent in many of the presentations that the nitrate losses recorded in mass balance calculations very rarely equate with direct measurements of denitrification. In discussions at the conference, and in some poster presentations (see Appendix I), it is apparent that our understanding of nitrate processing needs to be more sophisticated.

Microbiological communities may be involved in the assimilation of nitrate far in excess of their respiratory needs. This stored nitrogen may be incorporated into the denitrification process at a later stage, maybe when the water table lowers in the spring months and soils start to dry. There is a need to understand these retention mechanisms within saturated soils and also during wetting and drying sequences. If assimilation of nitrate-nitrogen does occur, the eventual prospect of N-saturation of buffer soils or sediments becomes real (Groffman, this volume).

The lessons learnt about nitrogen seem to apply equally well to phosphorus. As a consequence, a number of questions arise. If the microbiological pool does become N or P saturated, what influence will above-ground biomass management have on the dynamics of the microbiological community? Are there other functions that the buffer zone still undertakes? Does the environment still process inorganic nutrients, converting them into organic complexes which are then subsequently released to the riverine environment? What is the fate of these N- and P-rich organic complexes and are they in themselves an important product of these environments?

HYDROLOGICAL CONSIDERATIONS

Many of the processes occurring within buffer zones are highly dependent on hydrological, hydraulic and hydrogeological factors (Burt, this volume). The retention of nitrogen and phosphorus within stream environments is a case in point (Hill, this volume). Hill demonstrates that denitrification, and also sediment-P retention, are critically dependent on the water residence time within the stream complex. Since flow velocity influences the macrophytic species composition of a stream or river, flow can have a secondary impact on nutrient retention. Hill demonstrates that certain types of macrophyte can stimulate sediment denitrification through the combined processes of organic-nitrogen deposition and then nitrification. The stream environment, particularly during low flows, can therefore become an important processing environment for nitrogen and phosphorus.

The importance of hydraulic loading in relation to the buffering capacity of ponds and wetlands in the Swedish landscape was reported by Fleischer *et al.* (this volume). Two issues arise out of this work. Firstly, there is a limit to the catchment size that a pond or wetland can buffer before the hydrological loading fundamentally compromises key physical and biological processes in the buffer. Ponds or wetlands need to be placed strategically within a landscape. The second point relates to the degree of hydrological connectivity between the buffer (in this case, ponds and wetlands) and the source of the pollution. The connection of ponds to main channels has become an integral part of the enhancement of nutrient retention within the intensive arable landscape of southern Sweden.

This topic was also raised by Johnston *et al.* (this volume) with regard to the natural functioning of riverine wetlands. They point out that the natural connectivity between rivers and their floodplains has been lost in the vast majority of environments. The fluctuation of river levels is critical to the biological and physical processing of both nutrients and carbon within a wetland ecosystem. The loss of these natural fluctuations must have an impact on the dynamics of any ecosystem as well as on its structural diversity. Dam construction, channelisation and a vast array of engineering projects that either directly or indirectly influence the natural fluctuations in river levels all have an impact on the functioning of riverine wetlands. The question is whether these engineering works have a significant impact on the processing of nutrients within the wetland. Major changes in river-wetland dynamics, such as the regulation of downstream flow rates and river levels by dams, have major ramifications for riverine wetland and floodplain environments. Similarly, the loss of sediment loads in rivers due to dams has physical ramifications for the stability of floodplain banks and thus the nature and extent of wetland habitats. The downstream effects of a dam is illustrated well in Mana Pools wetlands along the Zambezi River, downstream of Lake Kariba (Zimbabwe/Zambia). The trapping of sediment in Lake Kariba provides a silt-free water, causing an increase in bank erosion below the dam. Furthermore, a decrease in flood peaks and flood duration has affected the richness and the biomass of the wetland vegetation. This has lead to an over-grazing of the vegetation by wildlife which, in turn, reduces the effectiveness of vegetation in preventing erosion (DuToit, 1984). As a consequence, the Zambezi is evolving into a single-thread channel through the Mana Pools RAMSAR area. Backwater pools and wetlands are becoming drier and do not receive their annual replenishment of water via flooding. Many of the pools have simply reverted to grassed areas, flooded for limited periods from local runoff.

For smaller wetland habitats (<1 km^2), the role of hydrology is no less important. However, there is greater potential to restore these habitats to act as buffer zones. The ability of a habitat to act as a sediment filter is discussed in detail by Dillaha and Inamdar (this volume). The delivery of sediments to the filter strip is critical. If flow is concentrated, one section of the buffer will receive a disproportionate amount of both water and sediment. The key is to locate a buffer, in this case a grass strip, sufficiently close to the source of the erosion so that flows are distributed across the whole strip, or to ensure that flow is spread over the buffer through the use of bunds (water bars) and flow diversion structures. Dillaha and Inamdar also point out the need to consider the medium- to long-term impact of sediment accumulation within a buffer. As more sediment is accumulated the profile of the buffer will change. In the case of a grass strip buffer, the leading edge of the buffer becomes higher, water may start to pond at the edge of the buffer and ultimately flow to the lowest point along this edge, thus concentrating water and sediment, leading to excessive flow into the buffer strip. Models which can simulate the effects of sediment accumulation are therefore important. A sediment buffer strip will have a finite life and will need to be managed. If one is using a pond or wetland then sedimentation will, in the long term, lead to terrestrialisation, unless the pond is dredged.

The structure of the floodplain soil and sediments, and the underlying geology, all influence how water moves through a riparian environment. Lowrance (this volume) discusses in detail the findings of the Chesapeake Bay Program which reviewed the potential of riparian habitats to buffer diffuse pollution on the basis of the macro-geological structure of the environment. In many cases, groundwater flows were considered to flow too deep to be effectively intercepted by either vegetation or microbiological communities. On the whole, too few studies have been done in "difficult" geological environments. As researchers we have tended to look at simple environments, with simple flow patterns, and have ended up with a set of studies that are somewhat similar. There is, therefore, an obvious need for research into the behaviour of buffer zones in different geologies, particularly where groundwater flow is complex. Modelling flow, dating water and field instrumentation become more demanding, but the assessment of these different geological settings is essential.

Hydrological processes control the processing of pollutants but many of the studies reviewed by this book's authors do not present complete hydrological budgets. In addition to looking at more diverse geological environments, there is a need to ensure hydrological budgets are complete so that chemical budgets are based on a sound foundation. In chemical budgets, there is a tendency to ignore soluble organic fractions of the nutrient budget. Many have assumed phosphorus to be transported only in overland flow, but as Mander *et al.* (this volume) have shown, significant quantities do exist in groundwater. How prevalent is this phenomenon?

HABITAT AND WATER QUALITY

The issue of balancing the conservation of an environment with its utilisation in buffering upstream impact was ever present in questions and workshop discussions. There are many risks inherent in using a habitat as a buffer and many habitats, in themselves, need to be protected. In the UK it may be somewhat naive to think that we would knowingly use the fragmented remains of our riparian wetlands to buffer large nitrogen or phosphorus loads and yet this is what often happens. In Europe as a whole, there is a need for large-scale creation of buffer zones. The situation may be somewhat different in the Eastern USA, where extensive areas of riparian wetlands, ponds etc. already exist. It is important to recognise that each habitat has a diversity of values and that in some cases it would not be appropriate to use the habitat as a buffer zone. A large-scale example of this is the case of the Danube Delta. Initial suggestions were made to use the delta wetland as a buffer for the Black Sea, thereby protecting the sea from the pollution within the River Danube. However, the percentage of flow through the delta wetland could only ever be increased from 3 to 5% and yet this marginal increase would fundamentally damage the biodiversity of the wetland.

Gumiero and Salmoiraghi (this volume) examined the invertebrate ecology of riparian zones. Their key conclusion was that in many of the sites they investigated the riparian zone vegetation made an active contribution to the ecology of the river itself, but in many cases the full potential of river and riparian interactions (as defined by invertebrate indices) was not realised. This results from the nature and extent of the buffer zone local to the study site and the extent of riparian vegetation along the river's length. In many cases, the nature of the riparian vegetation does not seem to have a significant impact on the environment's ability to control nitrate pollution. This has led to some very simple recommendations on the nature of vegetation within riparian zones. However, it must be remembered that these environments do have an impact on the ecology of rivers and all downstream habitats. Gumiero and Salmoiraghi's paper is an important reminder that buffer zone concepts are part of a broader objective which seeks to improve the quality of the catchment environment not only in terms of pollution abatement, but also in terms of the catchment's structural and ecological diversity.

STRATEGY FOR IMPLEMENTATION

Defining the mechanisms of how to implement the ideas enclosed in this book is an immense task. As Gardiner and Perala-Gardiner (this volume) point out, there is massive scope for the enhancement of river corridors but we also need the political will and commitment of the farming community. Cooper (this volume) observes that in New Zealand the farming community is already convinced of the off-site impacts of their activities and has developed a voluntary framework for the implementation of riparian buffers. Scientific findings are reviewed cooperatively with farmers in order to develop ideas for integrating riparian buffer strips into farmland and key concerns are addressed face to face with the farmer. For example, how should the buffer strip be managed and how does a change in the use of near-stream land influence the farmstead's economics? At a planning level, agencies need procedures to help answer these questions and decide which combination of measures is best suited to a particular farm.

Dickson and Schaeffer (this volume) also focus on the impact that a riparian buffer zone policy has on the farmstead. They examined the impact on critical farms next to streams margins in Illinois. Key concerns were the degree of fragmentation a farm would experience if riparian woodlands were planted, the effect of excessive fragmentation on the value of this ecotone and the influence of the land's rentable value on farmers' willingness to restore riparian woodland. Baudry (this volume) also comments that the financial dependency of a farmer on riparian land will fundamentally influence the degree to which a farmer will contemplate changes to this land. This dependency may take the form of the value of the land as pasture close to milking sheds or its role as summer grazing land. In intensively cultivated environments, Cook (this volume) notes that a buffer zone policy does have a considerable economic penalty, especially in areas with small fields and numerous ditches. Cook also notes that in some areas, the imposition of a buffer zone policy may have unplanned consequences, namely that a farmer may still want to keep the same proportion of the farm under arable cultivation; as a result the loss of land to riparian strips could be balanced by the removal of hedgerows and/or woodland within the farm. As Cook (this volume) and Dickson and Schaeffer (this volume) comment, there is a need to have whole-farm management plans, where the impacts of various options can be balanced against risks to both income and other habitats on the farm.

Riddell-Black *et al.* (this volume) suggest that buffer zones could be viewed as a curative approach. Put simply if a farmer implements buffer zones, it does not mean that farming can continue as normal. There is still the need to implement preventative measures. Dillaha and Inamdar (this volume) conclude that in-field Best Management Practices may hold sediment losses at acceptable levels but current Best Management Practices are limited in their ability to control chemical losses. There is a need to recognise that a buffer zone policy is part of a range of in-field and off-field management tools that can be deployed.

Communicating the ideas and principles of buffer zones to a farming/land management community should be relatively easy (see for example the UK's Environment Agency brochure: Buffer Strips, 1996). Tailoring the solution to the individual farm, however, and reviewing the economic as well as management impacts on the farm is more difficult. We are starting to see the emergence of procedures for the implementation of buffer zones (Collier *et al.*, 1995) but many of these rely on a large dose of "Best Professional Judgment" (Lowrance, this volume). Few studies have reported the impacts of riparian restoration from an ecological/water quality perspective, let alone the impact to the farm itself. Downes *et al.* (this volume) call for a period of research by management or the development of demonstration projects where the ecological and economic benefits, or otherwise, can be documented.

At a catchment scale, the issue of spatial priorities also needs to be determined. Diffuse pollution is not spread evenly throughout a catchment. Land use activities, soil types, geology, etc., all influence the nature and magnitude of diffuse pollution. It would seem obvious that some form of catchment risk assessment should be carried out as part of a planned implementation policy. This would identify specific farms that require a detailed whole-farm management plan, as well as critical stream reaches where survey work could be undertaken to determine the nature and extent of the impact. This hierarchical approach has been recently used by the UK Government's conservation agency, English Nature, on the River Lugg SSSI (Gibson, pers. comm.). Soil and topographic information were merged into a land vulnerability index and then critical reaches of the River Lugg were visited. As a result, sections of the river were prioritised for the establishment of buffer zone habitats and for associated grants. Simple field assessment techniques allowed local officers to develop solutions for buffer strips (mostly grass strips but with options for woodland). Farm visits, coupled with public meetings, helped communicate the ideas and raise the level of awareness of the role and importance of the riparian vegetation.

In the UK it is primarily the role of the Environment Agency and Ministry of Agriculture, Fisheries and Food (MAFF) to make the buffer zone concept a reality. MAFF have undertaken a five-year programme to test the feasibility of riparian habitat management. Tytherleigh (this volume) documents the aims and objectives of the UK's Water Fringe Option under the EU's Habitat Scheme (CEC 2078/92). The implementation of the scheme in the UK is based on a set of prescriptions applied to a number of trial rivers. Tytherleigh comments that the scheme has shown only modest success (12% of eligible land has been adopted in two years) since farmers are asked to sign a long-term agreement with no opt-out clause. The fragmentation of fields and, in some cases, the loss of stream-side watering of livestock has prevented many farmers from entering the scheme. There is a marked contrast between the UK's Water Fringe Option and New Zealand's implementation of riparian buffers. Both schemes are voluntary. The UK's scheme is prescriptive and tightly defined with little or no flexibility left to local offices whilst New Zealand's scheme is adaptable and appears to centre on the development of tailored assessments for each farm. New Zealand seem to be pushing for riparian woodland whilst in the UK's scheme the most popular option is grassland. Cooper (this volume) comments that New Zealand farmers recognise the impacts of their activities. It is questionable whether the UK farming community does, since they are unwilling to change land to a non-agricultural use, even with the presence of substantial subsidies. Lessons learnt from MAFF's Water Fringe Option and also the New Zealand experience have influenced the River Lugg Wildlife Enhancement Scheme (WES) and critical sections of the river have already been adopted into the scheme (Jennings, English Nature, pers comm).

Implementation of CEC 2078/92 could be regionally focused, as is happening in Tuscany, Italy (Cenni, Parco Naturale, pers. comm.), and the funds used to develop a more tailored scheme for the implementation of buffer zones. But maybe the chief obstacle to the implementation of buffer zones in Europe is a failure of the farming community to acknowledge the impact of its activities. Furthermore, the European Union's Common Agricultural Policy (CAP) seems so rigid in its drive for farm productivity that this has generated a socio-economic climate and dependency on high value, high income, high rents and high overhead cropping practices. As a consequence, even if farming impacts are recognised, it is difficult to change farming practices; agro-environmental schemes presented to farmers are always compared to the losses they will incur relative to the inflated prices of subsidies they currently get.

Gardiner and Perala-Gardiner (this volume) have outlined the principles of a sustainable restoration of buffer zones. They identify a need for: community involvement, skilled and willing people to manage the habitat and maintain its value and, most importantly, a willingness to change habits and attitudes and adjust to new financial circumstances. They argue that integrated catchment management, tied to clear objectives, is a credible vehicle to achieve this objective.

WHAT NEXT?

The lessons learnt from the various groups represented at the conference offer an optimistic view of the future of buffer zones. The consensus now is for the creation of demonstration projects that serve to educate people about the nature and value of buffer zones and in which new models and tools and specialist understanding can be developed further. The conference has, in addition, enabled us to identify a range of specific issues that need to be resolved if we are to take the concept of buffer zones forward. Here we summarise the key issues.

In relation to the processing of pollutants within buffer zones, the following questions remain:

- Do pesticides and, in particular, fungicides interfere with denitrification and chemical reduction processes?
- To what extent are N and P retention interdependent? Nitrogen and phosphorus retention do not seem to be maximised in the same environment. There are suggestions that whereas nitrogen retention is maximised in a wet soil environment, optimal phosphorus retention requires drier conditions. However, engineering a physical separation of the processes may prevent either occurring at optimal levels.

- What is the nature and source of the dissolved organic carbon (DOC) required for microbiological processes within a buffer zone and, in particular, within alluvial sediments where groundwater nitrate removal occurs. If the type of DOC is critical, what influence does above-ground vegetation have?
- What is the impact of water level and soil temperature fluctuations on buffer zone processes? In temperate environments, where most research has occurred, it is suggested that these fluctuations encourage carbon cycling and thus sustain a high potential for nitrogen retention within the buffer zone. However, we need to look at the dynamics of buffers in tropical environments, where soil temperatures and hydrological conditions are stable in order to compare their effectiveness.
- Measurements on denitrification in saturated zones often fall short of observed changes in groundwater quality. Is most of the processing of nutrients taking place in the unsaturated zone? What are the interactions between the processes occurring in the saturated and unsaturated zones?
- How effective are buffer zones in complex hydrogeological environments? Does the value of the concept hold for a more representative range of soil/geological catenas than have been studied to date.
- To what extent do buffer zones convert inorganic nutrients to organic compounds? There is a general need for the completion of both hydrological and nutrient budgets in studies, particularly with a focus on the dissolved organic fractions of N and P.
- What are the long-term benefits of buffer zones? We need to establish the initial water quality of a catchment, prior to experimentation or create a series of demonstration sites, and follow this by long-term monitoring.
- How do the different options compare, both between various buffer zone habitats and against the effectiveness of other practices (BMPs or COGAP)?

At both an operational and a strategic level, implementation does seem to depend on being able to define different objectives geographically within a catchment and target areas of priority. Tools are required to identify areas of diffuse pollution risk and to determine the range of buffer options available. These tools might include modelling routines based on GIS programmes or simple geographic indices derived from published maps.

When evaluating the suitability of a habitat to be a buffer zone, more locally applied tools are required to determine the nature of diffuse pollution and the performance or tolerance of the target habitat. Dillaha and Inamdar (this volume) cite work related to models on vegetated filter strips (VFS). These procedures build on existing erosion models and enable trained users to evaluate the potential of a given VFS to control sediment losses. In relation to nutrients we seem a long way from this level of sophistication. The riparian ecosystem management model (REMM; Lowrance, this volume) will allow a higher degree of assessment but needs further development, both in the USA and Europe, before being applied in a management or decision support context.

CONCLUSION

In our summing up of the conference we have deliberately not attempted to define the width, size or area required for a buffer zone, despite a general fixation with “how wide” the zone should be. The challenge for the scientific community is to engage policy and operational staff into the debate of how, why, where and what type of buffers could be created rather than taking the short-cut route of asking how wide. Given the complexity of buffer zone processes and their various purposes, a useful retort to the question of “how wide” is “how wide do you want it”!

Buffer zone habitats, whether they be ponds, wetlands, wet grassland, riparian woodland or hedgerows, do have multiple facets. Whether the benefits are through the creation of a diversity of catchment habitats, such as through shading of streams and thus cooling of stream temperature in the summer, or whether it is through the creation of a diverse stream invertebrate ecology by the contribution of leaves and twigs to the stream, our appreciation of these many facets is essential to the identification of appropriate buffer zone strategies.

Finally, we must remember that the buffer zone concept is only one of a range of tools that can be used to mitigate pollution. The best solution is likely to include a range of approaches. Planning the implementation of buffer zones does not mean solely focusing on a small area of land next to a key habitat (usually a stream). One cannot think small scale: the process involves assessing the whole catchment and all its farms in some detail.

REFERENCES

Collier K.J., Cooper A.B., Davies-Colley R.J., Rutherford J.C., Smith C.M. and Williamson R.B. (1995) *Managing Riparian Zones: A contribution to protecting New Zealand's rivers and streams. Volumes I & II.* NIWA, Department of Conservation, Science Publications, P.O. Box 10, 420 Wellington, New Zealand.

DuToit, R.F. (1984) Some environmental aspects of proposed hydroelectric schemes on the Zambezi River, Zimbabwe. *Biol. Conservation*, **28**, 73-87.

APPENDIX I

List of poster presentation authors, poster titles and the address of the principal author. Where known, email addresses are also given. Information as given in August 1996. The list is in alphabetical order relative to the surname of the first author.

Aguiar, F., **Riparian vegetation of the Sado River Basin (SW Portugal).** Instituto Superior de Agronomia, Universidade Tecnica de Lisboa, Seccao Autonoma de Arquitectura Paisagista, Tapada da Ajuda, 1399 Lisboa Codex, Portugal.

Badre, B., Nobelis, P., Sánchez-Pérez, J.M. and Trémolières, M., **Influence of degree of submersion on the decomposition rate of oak litter. Dynamics of the mass and chemical losses.** Laboratoire de Botanique et D'Ecologie Végetale, CEREG – URA 95 CNRS, Institute de Botanique, Universite Louis Pasteur, 28 Rue Goethe F-67083, Strasbourg Cedex, France (email:sanchez@equinoxe.u-strasbg.fr).

Butturini, A. and Sabater, F., **Effects of riparian vegetation removal on fluxes of dissolved matter in a mediterranean stream.** Departament D'Ecologia, Facultat de Biologia, Universitat de Barcelona, Avgda Diagonal 645, 08028 Barcelona, Spain (email:sabater@porthos.bio.ub.es).

Cacho, M., Radke, J.D. and Kondolf, G.M., **An Environmental Planning Model for the Design of Buffer Zones.** Department of Landscape Architecture, 202 Wurster Hall, Berkeley, CA., 94710, USA (email:cacho@ced.berkeley.edu).

Charter, E., **Potential for enhancement of a major trout fishing loch through creation of buffer zones.** FWAG Orkney, c/o SAC, 66 Junction Road, Kirkwall, KW15 1AR, UK.

Comin, F., **Restored freshwater marshes as buffer zones for nitrogen removal from agricultural runoff.** Department of Ecology, Faculty of Biology, University of Barcelona, Diagonal 645, Barcelona 08028, Spain (email:comin@porthos.bio.ub.es).

Dickson, B.C. and Schaeffer, D.J., **Ecorestoration of Riparian Forests for Non-point Source Pollution Control: Policy and Ecological Considerations in Agroecosystem Watersheds.** Confluence Ecological, P.O. Box 292, Reno, PA, 16343, USA (email:BCDICKSON@aol.com, bcd@galacticis.com).

Durand, P. and Merot, P., **Water pathways and the denitrification in the riparian zone: a catchment scale study.** U.R. de Sciences du Sol et Bioclimatologie, INRA-ENRA, 65 route de saint Brieuc, 35042 Rennes cedex, France (email:durand@roazhon.inra.fr).

Environment & Land Use Department, NFU, **Buffer zones – a farming viewpoint**. Environment and Land Use Department, National Farmers' Union of England and Wales, 164 Shaftesbury Avenue, London, WC2H 8HL, UK.

Franco, D., Perelli, M. and Scattolin, M., **Buffer strips to protect the Venice Lagoon from non-point source pollution.** Via Negroponte, 7C, Lido – Venezia (VE), 30126 , Italy (email:daniel.franco@iol.it).

Fuglsang, A., **Full scale example project about establishing wet meadows alongside waterways.** Department of Nature and Water Environment, Funen County Council, Ørbækvej 100, DK-5220 Odense Sö, Denmark (email:fynanv@inet.uni-c.dk).

Fuglsang, A., **Nutrient transformation in established uncultivated zones along watercourses.** See above.

Fustec, E., Fardeau, J.C., Bonté, P. and Engerran, F., **Short- and long-term accumulation and bioavailability of particulate phosphorus in flooded riparian zones in the Seine river basin (France).** Labaratoire de Geologie Appliquee, Universite P. et M. Curie, 4 Place Jussieu, T. 26, 75252, Paris Cedex 05, France.

Gibson, M., Jennings, S. and Haycock, N., **Protection of River SSSIs from Diffuse Pollution – a case study of the River Lugg Catchment.** English Nature, Masefield House, Wells Road, Malvern, WR14 4PA, UK.

Gril, J.J., Fagot, M. and Perret, I., **Grass buffer strips to limit contamination of freshwaters by pesticides research and actions in France.** CEMAGREF, 3 bis quai Chauveau CP110, F-69336, cedex 09, Lyon, France (email:jean-joel.gril@cemagref.fr).

Gumiero, B. and Salmoiraghi, G., **Relation between land use, river bank biodiversity and lotic macroinvertebrate community.** Department of Biology, University of Bologna, Via Selmi, 3, Bologna, 40126, Italy (email:peg0693@iperbole.bologna.it).

Hairsine, P., **Comparing grass filter strips and near-natural riparian zones for trapping sediment and sorbed nutrients.** CSIRO, Division of Soils and the Co-operative Research Centre for Catchment Hydrology, PO Box 639, Canberra 2601, Australia (email:peterh@cbr.soils.csiro.au).

Harris, G., Forster, A., Hollis, G., Mason, D.J. and House, W., **Particulate transport of pesticides – the potential for buffers to reduce contamination of surface waters.** ADAS, Anstey Hall, Maris Lane, Trumpington, Cambridge, CB2 2LF, UK.

Haygarth, P. and Jarvis, S., **Transfer of phosphorus from agricultural soils.** Institute of Grassland and Environmental Research, North Wyke, Okehampton, Devon, EX20 2SB, UK (email:haygarth@bbsrc.ac.uk).

Heppell, K., Burt, T.P. and Williams, R., **A consideration of the possible reductions in herbicide pollution from the use of buffer strips in structured clay soils.** School of Geography, Mansfield Road, Oxford, OX1 3TB, UK.

Hodum, O.S. and Burt, T.P., **Phosphorus distribution and speciation in riparian zone soils: implications for the design of buffer strips.** School of Geography, Mansfield Road, Oxford, OX1 3TB, UK.

Horwath, W.R., Griffith, S.M., Steiner, J.J., Elliott, L.F. and Wigington, P.J.(Jr)., **Denitrification Potential of a Natural Grass Riparian Zone and Adjacent Perennial Ryegrass Seed Production Field.** National Forage Seed Prod. Research Center, USDA-ARS , 3450 SW Campus Way , Corvallis, OR, 97331-7102, USA (email:griffits@ucs.orst.edu).

Hunt, P.G., Stone, K.C., Humenik, F.J. and Matheny, T.A., **Buffer Zones for the mitigation of nitrogen from a swine wastewater spray field.** Coastal Plains Research Center, USDA-ARS, 2611 W. Lucas St., Florence, South Carolina, 29501-1241, USA (email:hunt@florence.ars.usda.gov, hunt@sunbrn.florence.ars.usda.gov).

Inamdar, S.P. and Dillaha, T.A., **Hydrologic and sediment transport processes on riparian hillslopes.** USDA-ARS Southeast Watershed Research Lab, P.O. Box 946, Tifton, GA, 31793, USA (email:rivers@tifton.cpes.peachnet.edu).

Irvine, B.J. and Ray, D., **The effective management of sediment during upland afforestation.** Dept of Geography, Lancaster University, Bailrigg, Lancaster, LA1 4YB., UK (email:B.Irvine@lancaster.ac.uk).

Jenns, N.J. and Burt, T.P., **Floodplain and wetland sedimentation in a lowland agricultural catchment: mineral magnetics and caesium-137 as indices of topsoil delivery to riparian zones**. School of Geography, Mansfield Road, Oxford, OX1 3TB, UK (email:tim.burt@geography.oxford.ac.uk).

Kadikis, N. and Zandmane, A., **Trends in the ecological state of lakes in Kemeri (Latvia) wetlands as buffer zone**. Latvian Environment Data Centre (LEDC), 2, Straumes Str., Jumala LV 2015, Latvia.

Kasel, S. and Adams, M.A., **Soil N availability in relation to tree health and the role of riparian forests as buffer zones.** School of Botany, University of Melbourne, Parkville, Victoria, 3052, Australia (email:Ecology_Lab@muwayf.unimelb.edu.au).

Kluge W. and Schimming C.-G., **Hydrological and hydrochemical processes control the nutrient retention in an alder peatland of a lake shore.** Ecosystem Research Center, University of Kiel, Schauenburgerstr. 112, D-24118 Kiel, Germany (email:winfrid@pz-oekosys.uni-kiel.d400.de).

Leeds-Harrison, P.B., Walker, M.J. and Quinton, J.N., **Investigations of in-stream nitrate concentrations in a buffered clay catchment**. Department of Water Management, School of Agriculture, Food and Environment, Cranfield University, Silsoe, Bedfordshire, MK45 4DT, UK (email:stplh@silsoe.cranfield.ac.uk).

Maiolini, B., Boscaini, A., Del Prete, C. and Franceschini, A., **Carabid Communities in River Floodplains Along a Latitudinal Gradient Across Europe.** Musceo Tridentino di Scienze Naturali, Via Calepina 14, I-38100, Trento, Italy (email:maiolini@mtsn.itc.it).

Mander, Ü., Vought, L., Lõhmus, K., Kuusemets, V. and Ivask, M., **Nitrogen and Phosphorus Transformation in Riparian Alder Forests and Wet Meadows.** Institute of Geography, University of Tartu, Vanemuise Str. 46, EE-2400 Tartu, Estonia (email:ylo@math.ut.ee).

Matchett, L.S., Burt, T.P. and Goulding, K.W.T., **Denitrification in riparian buffer zones.** School of Geography, University of Oxford, Mansfield Road, Oxford, OX1 3TB, UK (email:lisa.matchett@geography.oxford.ac.uk).

Nunez Delgado, A., Lopez Periago, E. and Diaz-Fierros, F., **Effectiveness of buffer strips for attenuation of ammonium and nitrate levels in runoff from pasture amended with cattle slurry or inorganic fertiliser.** Dept. Edafoloxia e Quimica Agricola, Fac. Farmacia, Campus Univ S/N, 15706 Santiago de Compostela, Spain (email:edavelin@usc.es).

Olah, M., Toth, O.E., Olah, J. and Toth, J., **Denitrification capacity of gallery forests in a lowland river valley.** Ecological Institute of L. Kossuth University, H-4010, Debrecen, Hungary (email:TJA@tigris.klte.hu).

Patty, L. and Real, B., **Grassed buffer strips to reduce pesticide losses in runoff.** Institut Technique des Cereales et des Fourrages (ITCF), 107 Bd. de Stalingaad Bat. C, 69628 Villeurbanne Cedex, France (email:PATTY@lyon.cemagref.fr)

Price, D.A., Stewart, M.D., Bates, P.D., Anderson, M.G. and Burt, T.P., **An integrated hydraulic, hydrological and chemical model for floodplain prediction.** Department of Geography, Bristol University, University Road, Bristol, BS8, UK (email:price@gma.bris.ac.uk).

Prosser, I., Hairsine, P., Wilson, C. and Herron, N., **The Potential Role of Riparian Buffers for Improving Water Quality in Australia.** CRC for Catchment Hydrology, CSIRO Division of Water Resources, GPO Box 1666, Canberra, ACT, 2601, Australia (email:ianp@styx.cbr.dwr.csiro.au).

Puckett, L.J., Tornes, L.H., Cowdery, T.K., Brigham, M.E. and Stoner, J.D., **Transport and Fate of Nitrogen in the Riparian Zone: New Approaches Using Dissolved Gases and Chlorofluorocarbon Age Dating.** US Geological Survey, MS413, 12201 Sunrise Valley Dr., Reston, VA 22092, USA (email:lpuckett@er.usgs.gov, lpuckett@usgs.gov).

Quinn, P., Lord, E., Anthony, S. and Turner, S., **Strategic positioning of buffering zones within the landscape to minimise pollution losses and erosion.** ADAS Research and Development, Wergs Road, Woodthorne, Wolverhampton, WV6 8TQ, UK (email:Paul_Quinn@isuh.adas.co.uk).

Quinton, J.N. and Smith, R.E., **Simulation Analysis of Riparian Buffer Strips to Protect Surface Waters from sediment bound pollutants.** Department of Water Management, Cranfield University, Silsoe, Bedfordshire, MK45 4DT, UK (email:j.quinton@cranfield.ac.uk).

Riddell-Black, D., Mainstone, C.P. and Alker, G., **Economically viable buffer zones – the case for short rotational forest plantations.** European Topic Centre on Inland Waters ETC/IW, Water Research Centre – Medmenham, PO Box 16, Marlow, Bucks, SL7 2HD, UK (email:R_Black_D@wrcplc.co.uk).

Sabater, F., Butturini, A., Muñoz, I., Romani, A. and Sabater, S., **Buffer effects of riparian vegetation on instream processes.** Departament D'Ecologia, Universitat de Barcelona, Facultat de Biologia, Avgda Diagonal 645, 08028 Barcelona, Spain (email:sabater@porthos.bio.ub.es).

Sánchez-Pérez, J.M. and Trémolières, M., **Regulation of nutrient fluxes between surface water and groundwater in the alluvial forest ecosystems. Vertical component of the filtering effect.** Laboratoire de Botanique et d'Ecologie Végetale, CEREG – URA 95 CNRS, Institute de Botanique, Université Louis Pasteur, 28 Rue Goethe F-67083, Strasbourg Cedex, France (email:sanchez@equinoxe.u-strasbg.fr).

Sánchez-Pérez, J.M. and Trémolières, M., **Hydrological conditions controlling nutrient levels in groundwater in the alluvial forest ecosystems.** Laboratoire de Botanique et d'Ecologie Végetale, CEREG – URA 95 CNRS, Institut de Botanique, Université Louis Pasteur, 28 Rue Goethe F-67083, Strasbourg Cedex, France (email:sanchez@equinoxe.u-strasbg.fr).

Schnitzler, A., Sánchez-Pérez, J.M., Trémolières, M. and Weiss, D., **Impact of the floods on the riparian forest ecosystem's functioning. Modifications in the community structure, species composition and nutrient status.** Laboratoire de Botanique et d'Ecologie Végétale, CEREG – URA 95 CNRS, Institut de Botanique, Université Louis Pasteur, 28 Rue Goethe F-67083, Strasbourg Cedex, France (email:sanchez@equinoxe.u-strasbg.fr).

Schultz, R.C., Isenhart, T.M., Colletti, J.P. and Simpkins, W.W., **Integrated riparian management systems to protect water quality.** Department of Forestry, 251 Bessey Hall, Iowa State University, Ames, IA 50011, USA (email:rschultz@pop-3.iastate.edu, rschultz@iastate.edu).

Seymour, A.C. D'arcy, B.J. and Davies, D.H.K., **Buffer strip creation at West Fenton Farm, North Berwick-Scotland.** Scottish Environment Protection Agency, Clearwater House, Heriot-Watt Research Park, Avenue North, Riccarton, Edinburgh, EH14 4AP, UK.

Soulsby, C. and Chen M., **Groundwater abstraction from alluvial gravels in the riparian zone: a modelling study of river-aquifer interactions in the lower Spey, Scotland.** Department of Geography, University of Aberdeen, Elphinstone Road, Aberdeen, AB9 2UF, UK (email:c.soulsby@abdn.ac.uk).

Soulsby C. and Brogan J., **Riparian management on Scottish highland estates: an example of the potential and problems.** Department of Geography, University of Aberdeen, Elphinstone Road, Aberdeen, AB9 2UF, UK (email:c.soulsby@abdn.ac.uk).

Steiger J., **Retention of sediments during overbank flooding on the aquatic-terrestrial river ecotone.** Centre d'Ecologie des Systemes Fluviaux, CNRS , 29, rue Jeanne Marvig, 31055 Toulouse, France (email:steiger@cesf.cemes.fr, steiger@cemes.cemes.fr).

Stott, R.E., Burt, T.P., Rae, J.E. and Parker, A., **Potassium dynamics in riparian zone soils.** Postgraduate Research Institute for Sedimentology, University of Reading, Whiteknights, Reading, RG6 2AB, UK (email:R.E.Stott@reading.ac.uk).

Syversen, N. and Øygarden, L., **The use of tracer techniques (Cs^{134}) to study field erosion and sedimentation pattern within vegetative buffer strips (VBS).** Jordforsk, Centre for Soil and Environmental Research, 1432 As, Norway (email:nina.syversen@jordforsk.nlh.no).

Van Dijk, P.M. and Kwaad, F.J.P.M., **Effects of grass strips on sediment load and hydraulics of shallow flow.** University of Amsterdam, Laboratory of Physical Geography and Soil Science, Nieuwe Prinsengracht 130, 1018 VZ Amsterdam, The Netherlands (email:pvdijk@fgb.frw.uva.nl).

Walker, M.J., Quinton, J.N. and Leeds-Harrison, P.B., **A comparison of the effectiveness of established and newly established buffer zones in reducing nitrate concentrations reaching water courses.** Department of Water Management, School of Agriculture, Food and Environment, Cranfield University, Silsoe, Bedfordshire, MK45 4DT, UK (email:stplh@silsoe.cranfield.ac.uk).

Wantzen, K.M., **Environmental degradation of streams in Mato Grosso, Brazil.** Max Planck Institut für Limnologie, Arbeitsgruppe Tropenökologie, Postfach 165, 24302 Plön, Germany (email:wantzen@mpil-poen.mpg.d400.de).

Wei-Ning Xiang and Wright, C.A., **GIS-based riparian buffer analysis: injecting geographic information into landscape planning.** Geography and Earth Sciences, Univ. of North Carolina at Charlotte, Charlotte, NC, 28223, USA (email:wxiang@uncc.edu).

White, S.K., Cook, H.F. and Garraway, J.L., **Barrier strips and nitrate attenuation in groundwaters.** Wye College, Ashford, Kent, TN25 5AH, UK (email:S.White@wye.ac.uk).

Wigington, P.J., Griffith, S.M., Horwath, W.R., Steiner, J.J., Elliott, L.F., Field, J.A., Bottomley, P.J. and Baham, J.E., **Water Quality Function of Riparian Areas in Agricultural Landscapes of Western Oregon, USA.** National Forage Seed Prod. Research Center, USDA-ARS , 3450 SW Campus Way , Corvallis, OR, 97331-7102, USA (email:griffits@ucs.orst.edu).

Wilson, J.M. and Griffiths, H., **The potential for using ^{15}N natural abundance measurements to investigate nitrogen dynamics in agricultural catchments.** Dept. of Agricultural and Environmental Science, University of Newcastle upon Tyne, Ridley Building , Newcastle upon Tyne , NE1 7RU, UK (email:j.m.wilson@ncl.ac.uk).

Worrall, P., Haycock, N.E. and Cenni, M., **Lake Massaciuccoli: Mitigation of Agricultural Impacts.** Penny Anderson Associates, 52 Lower Lane, Chinley, Stockport, SK12 6BD, UK (email:paa@paa-ecol.demon.co.uk).

Young, S., Davies, D., Smith, K. and Harris, G., **The potential of vegetative buffer strips to control nutrient losses to aquatic systems, following fertiliser and slurry applications to grassland used for silage production.** Privet Cottage, Wybunbury Lane, Haymoor Green, Cheshire, CW5 7HH, UK.

Location Index

Organism Index

General Index