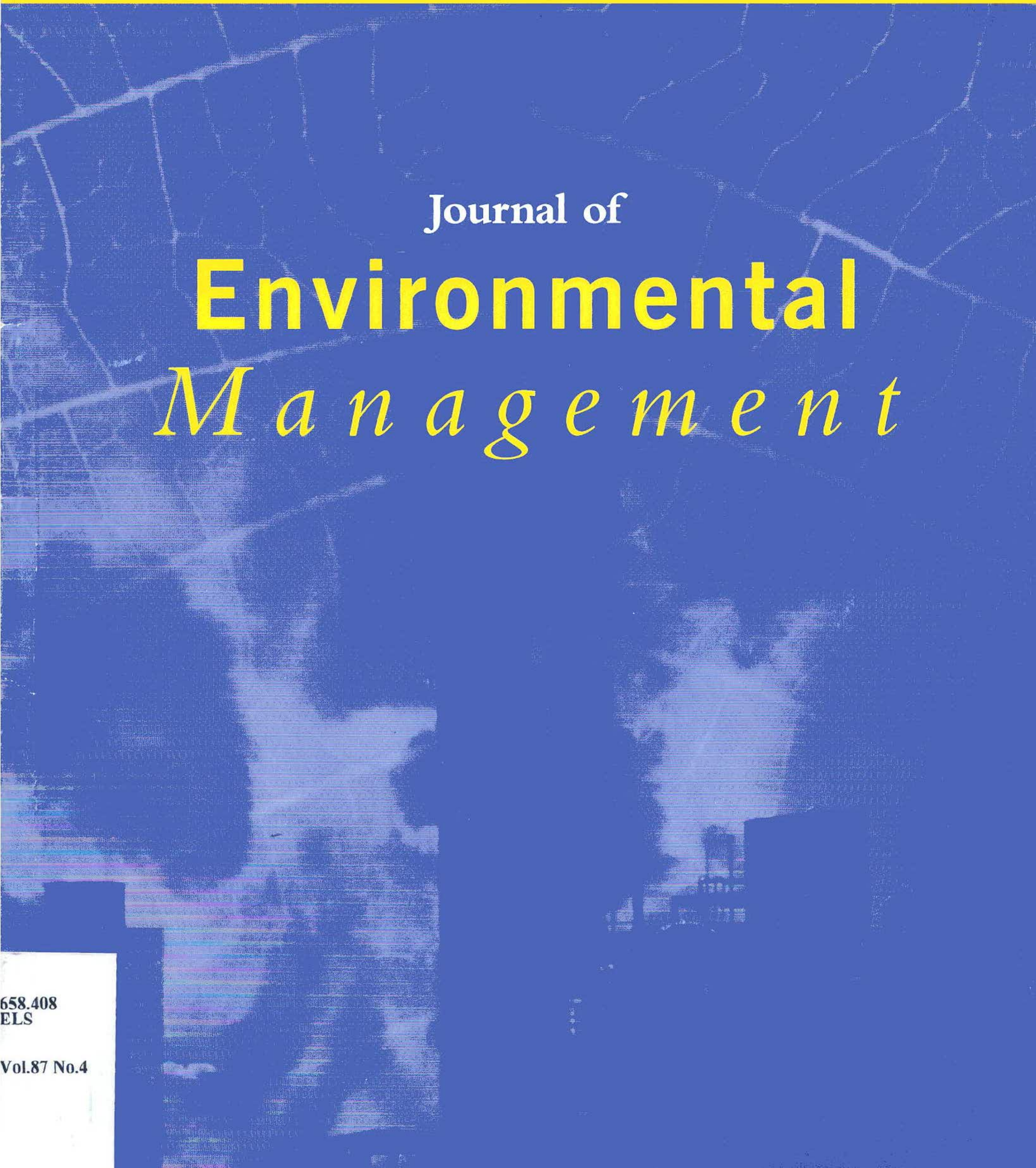
June 2008 Volume 87 Number 4 Special Issue

ELSEVIER



Journal of Environmental Management

Journal of

Environmental Management

A. GILL **Editor-in-Chief**
1620 Acton Street, Berkeley, CA 94702, USA.

Aims and Scope

The Journal of Environmental Management is a journal for the publication of peer reviewed, original research for all aspects of management and the managed use of the environment, both natural and man-made. Critical review articles are also welcome; submission of these is strongly encouraged.

As governments and the general public become more keenly aware of the critical issues arising from man's use of his environment, this journal provides a forum for the discussion of environmental problems around the world and for the presentation of management results. It is aimed not only at the environmental manager, but at anyone concerned with the sustainable use of environmental resources.

Research Areas Include, but are not exclusive to:

• resource quality, quantity and sustainability • economics of environmental management • transport and fate of pollutants in the environment • spill prevention and management • remediation of contaminated sites • process modification for pollution prevention • improved energy efficiency • waste treatment and disposal

Papers submitted should address environmental management issues using a range of techniques e.g. case studies, observational and theoretical analyses, the application of science, engineering and technology to questions of environmental concern or mathematical and computer modeling techniques with the aim of informing both the researcher and practitioner.

Note to contributors

Online submission is available for this journal, to submit your article online, go to http://ees.elsevier.com/jema. Instructions on how to submit a manuscript to the journal can be found in the Guide for Authors on the journal homepage at http://www.elsevier.com/locate/jenvman.

Journal of
Environmental
Management

Editor-in-Chief

A. GILL

Microbial and Nutrient Contaminants of Fresh and Coastal Waters

Guest Editor: Peter Hooda

ELSEVIER

Orders, claims, and journal enquiries: please contact the Customer Service Department at the Regional Sales Office nearest you:

Orlando: Elsevier, Customer Service Department, 6277 Sea Harbor Drive, Orlando, FL 32887-4800, USA; phone: (877) 8397126 [toll free within the USA]; (+1) (407) 5636022 [outside the USA]; fax: (+1) (407) 3631354; e-mail: JournalCustomerService-usa@elsevier.com

Amsterdam: Elsevier, Customer Service Department, PO Box 211, 1000 AE Amsterdam, The Netherlands; phone: (+31) (20) 4853757; fax: (+31) (20) 4853432; e-mail: nlinfo-f@elsevier.com

Tokyo: Elsevier, Customer Service Department, 4F Higashi-Azabu, 1-Chome Bldg, 1-9-15 Higashi-Azabu, Minato-ku, Tokyo 106-0044, Japan; phone: (+81) (3) 5561 5037; fax: (+81) (3) 5561 5047; e-mail: jp.info@elsevier.com

Singapore: Elsevier, Customer Service Department, 3 Killiney Road, #08-01 Winsland House I, Singapore 239519; phone: (+65) 63490222; fax: (+65) 67331510; e-mail: asianfo@elsevier.com

USA mailing notice: Journal of Environmental Management (ISSN 0301-4797) is published monthly, with additional copies in January, April, July, and October, by Elsevier Ltd. (The Boulevard, Langford Lane, Kidlington, Oxford OX5 1GB, UK). Annual subscription price in the USA US$ 1223 (valid in North, Central and South America), including air speed delivery. Periodical postage paid at Rahway NJ and additional mailing offices.

USA POSTMASTER: Send address changes to Journal of Environmental Management, Elsevier, 6277 Sea Harbor Drive, Orlando, FL 32887-4800

AIRFREIGHT AND MAILING in the USA by Mercury International Limited, 365, Blair Road, Avenel, NJ 07001.

Printed by the Alden Group, Oxford, UK

Author enquiries
For enquiries relating to the submission of articles (including electronic submission where available) please visit this journal's homepage at http://www.elsevier.com/locate/jenvman. You can track accepted articles at http://www.elsevier.com/trackarticle and set up e-mail alerts to inform you of when an article's status has changed. Also accessible from here is information on copyright, frequently asked questions and more.

Contact details for questions arising after acceptance of an article, especially those relating to proofs, will be provided by the publisher.

Funding body agreements and policies
Elsevier has established agreements and developed policies to allow authors who publish in Elsevier journals to comply with potential manuscript archiving requirements as specified as conditions of their grant awards. To learn more about existing agreements and policies please visit http://www.elsevier.com/fundingbodies

ELSEVIER

Journal of Environmental Management 87 (2008) 533–534

Journal of
Environmental
Management

www.elsevier.com/locate/jenvman

Editorial

Microbial and nutrient contaminants of fresh and coastal waters

Water quality protection is important from both human health and environmental perspectives. Drinking water quality is an issue of global human health concern, principally due to water contamination with pathogens and potentially toxic chemicals. Contaminants in freshwater resources can arise from a variety of sources including those associated with the treatment and disposal of sewage and agricultural livestock wastes. Pathogenic contaminants from these mixed sources include viruses and bacteria such as *Escherichia coli* O157I and the protozoa *Giardia* and *Cryptospordium parvum* raising issues in terms of drinking water (e.g., Directive 98/83/EC) and recreational use such as bathing waters (Bathing Water Directive, 76/160/EEC). For example, the town of Walkerton (Ontario, Canada) experienced the largest waterborne disease outbreak in Canada (Goss and Richards, 2008), which was linked to the contamination of the water supply system with pathogens that originated in manure, resulting in 2300 cases of gastroenteritis and seven deaths (cf. Goss and Richards, 2008). Similar outbreaks of cryptosporidiosis in England and Scotland were attributed to slurry spreading or other cattle activities (cf. Hooda et al. 2000a).

The Bathing Water Directive sets a number of mandatory microbiological and chemical standards which bathing waters should comply with. Within rural settings the major diffuse and point sources of microbial contamination are largely associated with livestock farming activities; however waters in catchments with no livestock or other apparent sources can be contaminated by wildlife sources (Hooda et al., 2000a). The compliance with microbiological standards can also be compromised in catchments with no obvious direct point or diffused sources, as discussed by Stapleton et al. (2008) where untreated sewage overflows accounted for $>90\%$ of the total microbial load. Clearly compliance with the Bathing Water Directive as well as for public health protection all potential sources need to be regularly assessed/ monitored in order to develop effective water quality management strategies.

Nutrient enrichment, mainly with phosphorus (P) of freshwaters can cause excessive growth of algae, leading to general degradation of aquatic ecosystems, including loss of biodiversity. The role of nitrogen (N) in eutrophication is less clear, however, there are other N related water quality issues, such as compliance with the nitrate-N limit (Nitrate Directive 91/676/EEC), and impacts of reduced-N on aquatic ecology. Excess inputs of either forms of the reduced N (NH_3–N or NH_4–N) can seriously impair aquatic ecology through direct toxicity (NH_3) and diminished dissolved oxygen (due to oxidation of NH_4) supply impacts upon invertebrates (Hooda et al., 2000b) and fish (Camargo and Alonso, 2006).

N and P arise generally from intensive agriculture (Hooda et al., 2000a; Withers and Lord, 2002); however, their inputs from sewage treatment works can be equally important in densely populated catchments (Jarvie et al., 2006). Within rural catchments, livestock farming presents more complex input sources of nutrients and carbon rich substances into waters. Among these sources is runoff from farmyards which can be highly contaminated with a variety of substances, including nutrients and faecal indicator organisms (Edwards et al., 2008). Similarly inputs of 'dirty-water' from animal housing, including milk parlour washings or overflow of silage effluents can seriously impair receiving waters by contributing a cocktail of contaminants. Such inputs include dissolved oxygen depleting labile carbon rich waste and large amount of nutrients (Edwards and Hooda, 2008). Direct discharges of such effluents from animal housing areas can also contribute significant quantities of faecal bacteria and NH_4–N (Monaghan and Smith, 2004). Together, these livestock farm effluents have been known to degrade ecological quality of receiving waters (Hooda et al., 2000b). Ecological status of waters has gained much more significance in recent years because of the Water Framework Directive (Directive 2000/60/EC), which places ecological quality protection at the centre of water management strategies.

The connectivity between fresh- and coastal-waters means contaminants arising from inland sources reach the marine environment. It is thus largely catchment sources, both rural and urban, that need to be targeted in order to control contaminants transport to coastal waters. As point sources are relatively easy to identify and control, the diffuse agriculture sources are becoming increasingly significant. As a result of this, farm nutrient management is becoming an important tool to reduce nutrient losses

doi:10.1016/j.jenvman.2007.08.020

to waters by implementing best management practices (Monaghan et al., 2008).

The special issue brings together current research that quantifies specific aspects of contamination and remediation of fresh and coastal waters. The paper topics have been selected to describe the issues, supporting research and finally management strategies. It is hoped that the issue makes a significant contribution to the understanding of microbial and nutrient contaminants for further improvement in water resource management, including ecological quality which is the mainstay in the Water Framework Directive.

References

Camargo, J.A., Alonso, Á., 2006. Ecological and toxicological effects of inorganic nitrogen pollution in aquatic ecosystems: A global assessment. Environment International 32, 831–849.

Edwards, A.C., Hooda, P.S., 2008. Farmyard point discharges and their influence on nutrient and labile carbon dynamics in a second order stream draining through a dairy unit. Journal of Environmental Management 87, 591–599.

Edwards, A.C., Kay, D., McDonald, A., Francis, C., Watkins, J., Wilkinson, J.R., Wyer, M.D., 2008. Farmyards, an overlooked source for highly contaminated runoff. Journal of Environmental Management 87, 551–559.

Goss, M., Richards, M., 2008. Development of a risk-based index for source water protection planning, which supports the reduction of pathogens from agricultural activity entering water resources. Journal of Environmental Management 87, 623–632.

Hooda, P.S., Edwards, A.C., Anderson, H.A., Miller, A., 2000a. A review of catchment water quality concerns in livestock farming areas. The Science of the Total Environment 250, 143–167.

Hooda, P.S., Moynagh, M., Svoboda, I., Miller, A., 2000b. Macroinvertebrate as bioindicators of water pollution in streams draining dairy farm catchments. Chemistry and Ecology 17, 17–30.

Jarvie, H.P., Neal, C., Withers, P.J.A., 2006. Sewage-effluent phosphorus: A greater risk to river eutrophication than agricultural phosphorus?. The Science of Total Environment 360, 246–253.

Monaghan, R.M., Smith, L.C., 2004. Minimising surface water pollution resulting from farm-dairy effluent application to mole-pipe drained soils. II. The contribution of preferential flow of effluent to whole-farm pollutant losses in subsurface drainage from a West Otago dairy farm. New Zealand Journal of Agricultural Research 47, 417–428.

Monaghan, R.M., de Klein, C.A.M., Muirhead, R.W., 2008. Prioritisation of farm scale remediation efforts for reducing losses of nutrients and faecal indicator organisms to waterways: a case study of New Zealand dairy farming. Journal of Environmental Management 87, 609–622.

Stapleton, C.M., Wyer, M.D., Crowther, J., McDonald, A.T., Kay, D., Greaves, J., Wither, A., Watkins, J., Francis, C., Humphrey, N., Bradford, M., 2008. Quantitative catchment profiling to apportion faecal indicator organism budgets for the Ribble system, the UK's sentinel drainage basin for Water Framework Directive research. Journal of Environmental Management 87, 535–550.

Withers, P.J.A., Lord, E.I., 2002. Agricultural nutrient inputs to rivers and groundwaters in the UK: policy, environmental management and research needs. The Science of the Total Environment 282, 9–24.

P.S. Hooda
Centre for Earth and Environmental Science Research, Kingston University London, Kingston upon Thames KT1 2EE, UK
E-mail address: p.hooda@kingston.ac.uk
25 July 2007

ELSEVIER

Journal of Environmental Management 87 (2008) 535–550

Journal of
Environmental
Management

www.elsevier.com/locate/jenvman

Quantitative catchment profiling to apportion faecal indicator organism budgets for the Ribble system, the UK's sentinel drainage basin for Water Framework Directive research

C.M. Stapleton[a], M.D. Wyer[a], J. Crowther[a], A.T. McDonald[b], D. Kay[a,*], J. Greaves[c], A. Wither[c], J. Watkins[d], C. Francis[d], N. Humphrey[a], M. Bradford[a]

[a] *Centre for Catchment to Coastal Research, River Basin Dynamics and Hydrology Research Group, IGES, University of Wales, Aberystwyth, SY23 3DB, UK*
[b] *Earth and Biosphere Institute, School of Geography, University of Leeds, Leeds LS2 9JT, UK*
[c] *Environment Agency North West Region, Richard Fairclough House Warrington, WA4 1HT, UK*
[d] *CREH Analytical, Hoyland House Leeds, LS18 4RS, UK*

Received 8 August 2006; received in revised form 26 October 2006; accepted 1 November 2006

Abstract

Under the EU Water Framework Directive (WFD) 20/60/EC and the US Federal Water Pollution Control Act 2002 management of water quality within river drainage basins has shifted from traditional point-source control to a holistic approach whereby the overall contribution of point and diffuse sources of pollutants has to be considered. Consequently, there is a requirement to undertake source-apportionment studies of pollutant fluxes within catchments. The inclusion of the Bathing Water Directive (BWD), under the list of 'protected areas' in the WFD places a requirement to control sources of faecal indicator organisms within catchments in order to achieve the objectives of both the BWD (and its revision - 2006/7/EC) and the WFD. This study was therefore initiated to quantify catchment-derived fluxes of faecal indicator compliance parameters originating from both point and diffuse sources.

The Ribble drainage basin is the single UK sentinel WFD research catchment and discharges to the south of the Fylde coast, which includes a number of high profile, historically non-compliant, bathing waters. Faecal indicator concentrations (faecal coliform concentrations are reported herein) were measured at 41 riverine locations, the 15 largest wastewater treatment works (WwTWs) and 15 combined sewer overflows (CSOs) across the Ribble basin over a 44-day period during the 2002 bathing season. The sampling programme included targeting rainfall-induced high flow events and sample results were categorised as either base flow or high flow. At the riverine sites, geometric mean faecal coliform concentrations showed statistically significant elevation at high flow compared to base flow. The resultant faecal coliform flux estimates revealed that over 90% of the total organism load to the Ribble Estuary was discharged by sewage related sources during high flow events. These sewage sources were largely related to the urban areas to the south and east of the Ribble basin, with over half the load associated with the relatively small subcatchment of the River Douglas. The majority of this load was attributed to two WwTWs that discharge through a common outfall close to the tidal limit of this catchment. Budgets adjusted to accommodate the impact of proposed UV disinfection of these effluents showed that the load from these sources would be reduced significantly during base flow conditions. However, during high flow events loads would still remain high due to the operation of storm sewage overflows from stormwater retention tanks. The study identified untreated storm sewage spills from urban infrastructure and WwTW stormwater retention tanks as the dominant component of the high flow flux of faecal indicators to receiving waters of the Fylde coast and the associated bathing waters.

Keywords: Water Framework Directive; Faecal indicators; Coliform; Enterococci; Diffuse pollution; Sewage; Drainage basin; Source-apportionment; Catchment profiling; Intermittent discharges

---

*Corresponding author. Tel./fax: +44 1570 423565.
E-mail address: dvk@aber.ac.uk (D. Kay).

0301-4797/$ - see front matter
doi:10.1016/j.jenvman.2006.11.035

1. Introduction

The introduction of the EU Water Framework Directive (WFD 20/60/EC (CEC, 2000)) represents a major shift from traditional point-source effluent quality regulation towards a catchment-wide approach (Kay et al., 2006a). This requires the consideration of point and diffuse pollution sources in concert in order to effect control of water quality at the point where it is used for eco-system maintenance, water supply, recreation or fisheries. Regulation of water quality under the Directive is based on the 'drainage basin' within which Member States are required to achieve 'good ecological status and quality' in their watercourses by 2015.

Crucially, Article 6 of the WFD requires the EU member states to identify 'protected areas' that 'have been designated as requiring special protection under specific Community legislation for the protection of their surface water and ground water or for the conservation of habitats or species directly depending on water' (CEC, 2000). The specific legislation referred to is listed in Annex VI of the WFD and includes, *inter alia*, areas covered by the Bathing Water Directive (BWD 76/160/EC (CEC, 1976)). The principal compliance parameters of the BWD are faecal indicator organisms, i.e. total coliforms, faecal coliforms/*Escherichia coli* and intestinal enterococci, thus placing a legally binding obligation on the competent authorities in EU member states to limit indicator bacterial concentrations in bathing waters through a combination of point and diffuse source control (WFD Article 10). Article 11 of the WFD outlines the requirement for a 'program of measures' to implement the Directive with 'basic measures' being those necessary to achieve the criteria specified in the Directives outlined in Annex VI.

The recently agreed revision to the BWD (CEC, 2006) clearly states the expectation that the WFD will provide the tools for the implementation of the new 'health-evidence-based' microbiological guidelines for recreational waters. The basis for these standards was taken from the microbiological guidelines for recreational waters outlined in WHO (2003) and Kay et al. (1994, 2004, 2006b) whilst the Directive also promotes the concepts of the 'drainage basin-scale profile' of faecal indicator sources as recommended by WHO (1999, 2003) (namely; the 'Annapolis Protocol').

Similar principles to the WFD's catchment-wide management of water quality are contained within the USA's Federal Water Pollution Control Act (Anon, 2002), which uses a 'total maximum daily load (TMDL)' approach (Kay et al., 2006c). The Act requires that States identify water bodies that do not meet defined water quality standards and specify how much of a pollutant a water body can tolerate whilst complying with water quality standards. The TMDL investigation aims to quantify all pollutant fluxes from point sources such as wastewater treatment works (WwTWs) and diffuse sources from both agriculture and surface water drainage of urban areas (Anon, 2005). Once the contribution of the various sources have been determined remediation measures can be designed to reduce the pollution load within the TMDL, whether this is through management of point sources (e.g. improved sewage treatment) and/or diffuse sources (e.g. better agricultural management practices).

Key to the successful application of water quality management both in Europe and the USA, therefore, is the apportionment of pollutant fluxes to various sources identified within a catchment. In the UK, whilst monitoring and modelling of pollutants such as the nutrients, nitrogen and phosphorus are relatively well established (Defra, 2003; Haygarth et al., 2005), very little attention has been given to the monitoring and modelling of faecal indicator organisms in river catchments. Thus, there is a lack of empirical data on which to base initial assessments of faecal indicator loads (Kay et al., 2007a).

The study of the Ribble catchment described below is the first large-scale (area: 1583 km^2) catchment investigation of source-apportionment for faecal indicator fluxes in the UK, concentrating on the UK's sentinel drainage basin chosen to test the implementation and application of the new WFD principles. The study had two main aims: (i) to provide an assessment of point and diffuse sources of faecal indicators within the catchment; and (ii) to explore the relationships between remotely-sensed digital land cover data and water quality and to develop a modelling tool capable of predicting faecal indicator concentrations in watercourses without the need for extensive empirical data collection. This paper describes the empirical data collection exercise and resultant faecal indicator organism budgets, providing an assessment of source-apportionment within the River Ribble basin. The use of digital land cover data for the prediction of faecal indicator organism concentrations in the River Ribble basin, which utilised the water quality data described herein, is outlined in Kay et al. (2005a).

2. Materials and methods

2.1. Study area

The Ribble Estuary is located immediately to the south of the Fylde coast, Lancashire in the north west of England. This stretch of coast includes some of the UK's premier resorts and bathing beaches and has been subject to significant infrastructure investments (>£600 m) on point-source remediation to achieve the standards of the 1976 BWD. Five main subcatchments drain into the Ribble Estuary: the River Ribble, the River Darwen, the River Douglas, the River Lostock and the River Yarrow (Fig. 1). The largest subcatchment (1130 km^2) is drained by the River Ribble itself, which flows south from the flanks of the Pennine hills. Two major subcatchments of the Ribble, the River Hodder and the River Calder, drain from the north and south,

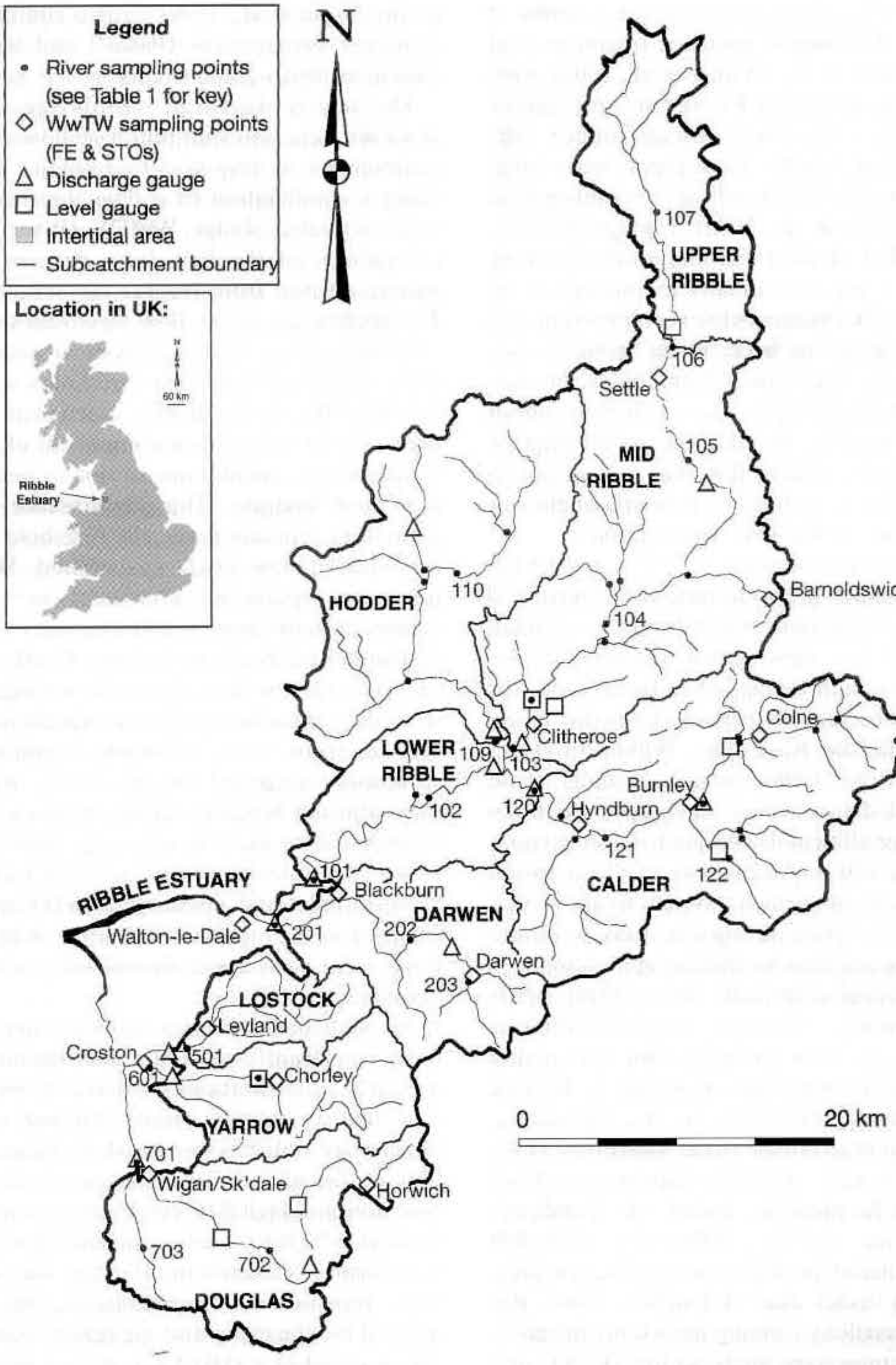


Fig. 1. The Ribble catchment showing the riverine and wastewater treatment works sampling sites used during the 2002 empirical data collection exercise.

respectively, joining with the River Ribble south of Clitheroe. The river then drains southwest through an area of lower relief to the tidal limit near Preston. The remaining four major rivers drain an area of comparatively low relief in the southern section of the river basin (Fig. 1). The subcatchments of the Rivers Calder, Darwen and Douglas contain extensive conurbations. Land cover in the remaining area is largely pastoral with some forestry in the upper reaches of the River Hodder subcatchment.

2.2. Field study design

Forty-one river monitoring points were selected from the Environment Agency (EA) sampling network, which included several flow gauge locations (Fig. 1). River flows were characterised from hourly discharge ($m^3 s^{-1}$) or stage (m) time series records from 22 hydrometric monitoring stations within the catchment operated by the Environment Agency (Fig. 1).

The 15 largest WwTW inputs, characterised in terms of treated final effluent (FE) faecal indicator organism load during a preliminary desk study (Wither et al., 2005), were selected for monitoring of treated FE and storm overflow effluent (i.e. overflows from storm storage and/or inlet combined sewer overflows, CSOs). These inputs represented over 95% of the estimated faecal indicator organism load from FE sources (Wither et al., 2005). Flow monitoring equipment was installed at most FE and storm overflow monitoring points if it was not already monitored on an hourly basis. One WwTW (Skelmersdale) comprised of two distinct treatment streams that necessitated sampling two FEs prior to their mixing. The effluent from this facility was discharged into the River Douglas via a common outfall shared with Wigan WwTW. In addition, monitoring of intermittent storm sewage effluent flow was carried out at 15 CSOs, one chosen from each sewer network catchment area associated with the 15 WwTWs selected above.

Sampling was undertaken between 30/7/02 and 10/9/02. Routine samples were taken at two to three day intervals to characterise base flow conditions. Opportunistic sampling in response to rainfall was implemented to target riverine sampling during hydrograph events, when faecal indicator organisms have been observed to increase (Crowther et al., 2002, 2003; McDonald and Kay, 1981; Wilkinson et al., 2006; Wyer et al., 1996, 1998). Sampling under these conditions also included stormwater surcharged FE flows and intermittent sewage effluent discharges from storm tank overflows (STOs) and CSOs. Access was provided to the EA flow monitoring station telemetry system to allow real-time monitoring of river levels during the study to ensure the optimum sampling response to hydrograph events.

Samples were collected aseptically into 150 ml sterile disposable plastic containers. A larger volume (250 ml) was collected at one site on each sampling run for quality control duplicate analysis. Samples were stored in the dark inside a cool box during transport to the laboratory. Indicator organism enumerations (total coliforms (TC), faecal coliforms (FC) and intestinal enterococci (EN)) followed standard UK methods based on membrane filtration (Environment Agency, 2000). FC and EN enumerations were filtered in triplicate to enhance measurement precision (Fleisher and McFadden, 1980). Results were expressed as colony forming units (cfu) $100\,\mathrm{ml}^{-1}$.

A total of 1299 samples were analysed for TC, FC and EN although 63 TC results (one for each sample site) were disregarded due to an incubator malfunction. Paired t-tests were used to compare duplicate bacteriological analyses of control samples with corresponding sample results. The results showed no significant differences (i.e. $p > 0.05$) for all three bacterial parameters (TC: $t = -1.23$, $p = 0.24$; FC: $t = 1.33$, $p = 0.20$; EN: $t = -1.14$, $p = 0.27$).

2.3. Data analysis and flux estimation

Hourly flow records for the riverine sites were separated into base flow and rainfall-induced high flow hydrograph events (Wyer et al., 1996) using a combination of bespoke computer programmes (Pascal) and visual inspection of individual hydrograph events.

The hourly discharge records for the FE at the 15 WwTWs were also split into base flow and high flow event components in response to rainfall. This was achieved using a combination of a flow threshold derived from a large activated sludge WwTW (Wyer et al., 1998) and comparison of the flow trace with a typical daily flow pattern derived from records for periods with no rainfall. The application of the flow threshold to other WwTWs is uncertain and a similar analysis of data from the current study suggested that such thresholds were not present at the WwTWs in the Ribble catchment. However, it was necessary for a high flow component of FE to be included in the budget calculations so not to unduly bias base and high flow budgets. Thus, in absence of an alternative method to separate flows, the threshold of 1.74 × average dry weather flow (dwf) was utilised. Whilst this method produced separations consistent with similar previous studies in most cases, it was necessary to employ a degree of manual intervention at some locations. In these cases, each record was then inspected for significant deviations from the typical flow pattern corresponding to rainfall data derived from local continuous rainfall recorders. The separation suggested by the 1.74 × dwf algorithm was then adjusted better to reflect increased flow derived from rainfall and to exclude any 'high flow' periods above the 1.74 × dwf threshold not associated with rainfall.

All samples and discharge at STO and CSO sites were assigned to the high flow category, as in normal operation flows from such assets should only occur during rainfall driven events.

All statistical analyses reported here were conducted using $\log_{10}$ transformed data since the distribution of faecal indicator concentrations showed a closer approximation to a normal probability density function when transformed. Descriptive statistics were used to characterise the distribution of faecal indicator concentrations collected during base flow and high flow conditions, including the geometric mean (GM) (calculated as the antilog of the mean of $\log_{10}$ transformed concentrations), the standard deviation of $\log_{10}$ transformed concentrations, the 95% confidence interval for the mean and the range. Statistical significance was assessed at $\alpha = 0.05$ (i.e. 95% confidence level or 5% significance level).

Faecal indicator organism flux under base flow and high flow conditions was estimated as the product of discharge and faecal indicator concentration for each site, i.e. the base flow and high flow geometric mean faecal indicator concentration for each source (i) was multiplied by the appropriate base flow (b) and high flow (h) discharge volume for the study period (9am GMT 29/7/02 to 9am GMT 11/9/02; 1056 h). The total load for each budget was calculated as the sum of the loads from each source contributing to the point on the river network for which the budget is being calculated (e.g. tidal limit).

Due to the large amount of information generated by this project, results are presented only for strategic locations mentioned in the text. Faecal coliform results are presented throughout as an exemplar given that distributions were similar for all three organisms. Broadly, TC concentrations tended to be an order of magnitude higher than FC concentrations that, likewise, were an order of magnitude higher than EN concentrations. Where variations were present, these are described in the text. A description of land use impacts on riverine faecal indicator organism concentrations is given in Kay et al. (2005a).

3. Results

3.1. Discharge during the study period

To contextualise flow conditions encountered during the field study an analysis of return periods was undertaken. Peak flow in the River Ribble at Samlesbury (657 $m^3 s^{-1}$) on 2/8/02 had a return period of one in seven years based on an annual series and 1 in 38 years based on the bathing season (1st May to 30th September). Peak flow in the River Darwen at Blue Bridge (84 $m^3 s^{-1}$, 2/8/02) had a bathing season series return period of one in four years. The peak flow in the River Douglas on 2/8/02 was not extreme and was lower than during later events. These differences in discharge magnitude can by accounted for by the difference in catchment topography, location and localised nature of summer storms.

Daily mean discharge (DMD) at the Samlesbury gauge for the 44-day study period was also compared with corresponding data for the same period during the previous 11 years (i.e. 1991–2001), using median values and non-parametric statistical analysis (Sign test of the median) due to the skewed distribution of values even following transformation. The results are shown in Fig. 2 as the median DMD and range for each year and the overall median and range for all years. The overall maximum value (343.87 $m^3 s^{-1}$) was recorded during the study period. However, the median daily mean discharge during the study period (13.51 $m^3 s^{-1}$) was not the highest recorded, which was 34.33 $m^3 s^{-1}$ recorded for the same period in 1998. The median DMD during the study was 1.76 times higher than the overall median value (7.68 $m^3 s^{-1}$). The results suggested that the median DMD during the study period was significantly elevated ($p = 0.0048$) compared to the overall value.

This analysis suggests that conditions during the study period were to some extent atypical with generally elevated discharge, indicated by the elevated median value, and extreme elevation in terms of the maximum value. The maximum was associated with localised, heavy and prolonged rainfall during the first week of the study which was observed to produce flood conditions in the upper Ribble catchment. Thus, conditions during the study may represent, to some extent, a case of elevated summer discharge.

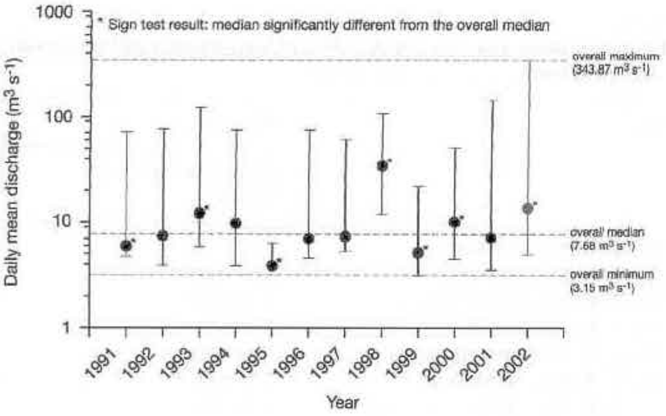


Fig. 2. Median and range of daily mean discharge ($m^3 s^{-1}$) at the Samlesbury gauging station on the River Ribble during a 44-day period corresponding to the 2002 field study (29 July to 10 September) for the years 1991–2002.

3.2. Faecal indicator organism concentrations at riverine sites

Faecal coliform results for key river and stream sample points are summarised in Table 1. At least six high flow samples were obtained from each site during the course of the study with ten or more high flow samples taken at 21 of the 41 sites. The results show wide temporal variations in faecal indicator organism concentrations and a statistically significant increase in concentration in response to hydrograph events. In most cases, the elevation in GM concentration at high flow was at least an order of magnitude over the base flow concentration.

The greatest base flow and high flow concentrations of all three faecal indicator organisms were found at Wanes Blades Bridge on the River Douglas (site 701) whilst the lowest concentrations were associated with rural headwater and tributary streams. Along the course of the River Ribble itself, the base flow GM concentration varies in response to inputs from WwTWs (e.g. between site 106 and 105 (Settle WwTWs) and sites 104 and 103 (Clitheroe WwTWs)). Concentrations remain similar in the lower catchment (sites 101–103) despite the input of water with relatively high GM concentrations from the River Calder (site 120), probably due to the input of cleaner water from the River Hodder (site 109) plus die off and sedimentation of organisms in the lower catchment. The River Calder shows a general increase in GM concentrations towards the catchment outlet at Whalley Weir (site 120) through inputs from WwTWs at Colne, Burnley and Hyndburn. The River Hodder has no substantial sewage effluent inputs and consequently has much lower GM FC concentrations. The catchments of the Rivers Darwen and Douglas, show pronounced increases in base flow GM concentrations towards the catchment outlets (sites 201 – Blue Bridge and 701 – Wanes Blades Bridge) relating to WwTW effluent discharges from the Darwen and Blackburn WwTWs (Darwen

Table 1
Geometric mean and standard deviation of $\log_{10}$ transformed faecal coliform concentrations (cfu 100 ml^{-1}) at selected river sampling points in the Ribble catchment between 30/7/02 and 10/9/02

Site	Base flow			High flow		
	Geometric mean (cfu 100 ml^{-1})	S.D.[a]	n[b]	Geometric mean (cfu 100 ml^{-1})	S.D.[a]	n[b]
101 R. Ribble, Samlesbury	5.57×10^3	0.325	11	4.98×10^{4c}	0.899	12
102 R. Ribble, Ribchester	1.30×10^4	0.364	10	1.25×10^{5c}	0.588	13
103 R. Ribble, Great Mitton	1.30×10^4	0.229	11	7.85×10^{4c}	0.682	12
104 R. Ribble, Sawley	3.74×10^3	0.296	12	7.57×10^{4c}	0.752	10
105 R. Ribble, Cow Bridge	3.40×10^3	0.373	12	3.66×10^{4c}	0.448	11
106 R. Ribble, Settle Weir	2.56×10^3	0.333	12	1.86×10^{4c}	0.426	10
107 R. Ribble, H'ton-in-Rib'dale	1.64×10^3	0.370	12	1.69×10^{4c}	0.514	10
109 R. Hodder, Lwr Hodder Br.	1.50×10^3	0.274	13	9.40×10^{4c}	0.489	10
110 R. Hodder, K'mere Man.	2.38×10^3	0.144	13	5.01×10^{4c}	0.647	7
120 R. Calder, Whalley Weir	5.15×10^4	0.458	11	2.31×10^{5c}	0.424	14
121 R. Calder, Atham Bridge	2.14×10^4	0.546	14	4.54×10^{5c}	0.374	9
122 R. Calder, Townley Park	3.50×10^3	0.307	13	4.09×10^{4c}	0.554	9
201 R. Darwen, Blue Bridge	5.75×10^4	0.501	11	9.27×10^{5c}	0.974	10
202 R. Darwen, Pl'ton Fields	1.82×10^4	0.477	11	3.48×10^{5c}	0.793	11
203 R. Darwen, Hollins Mill	1.58×10^4	0.378	11	9.33×10^{4c}	0.545	11
501 R. Lostock	1.60×10^4	0.308	10	1.95×10^{5c}	0.366	13
601 R. Yarrow, Fishery Bridge	2.61×10^4	0.254	12	2.99×10^{5c}	0.439	12
701 R. Douglas, Wanes Blades	5.37×10^5	0.794	12	3.57×10^{6c}	0.383	15
702 R. Douglas, Martland Bridge	1.69×10^4	0.276	13	3.49×10^{5c}	0.458	8
703 R. Tawd	3.33×10^4	0.420	13	1.64×10^{5c}	0.308	10

[a] S.D. = standard deviation of $\log_{10}$ transformed concentrations.
[b] n = number of observations.
[c] Results of Student's t-test show a significant difference in geometric mean at high flow compared to base flow at $\alpha = 0.05$ (95% confidence).

catchment) and Wigan and Skelmersdale WwTWs (Douglas catchment).

There is less distinct variation in high flow GM concentrations, possibly due to sewage effluent related inputs dominating under base flow conditions and diffuse catchment sources being mobilized during high flow response to rainfall. The River Ribble itself shows a general increase in high flow GM concentration downstream from Horton-in-Ribblesdale (site 107) to Ribchester (site 102). Relatively high GM faecal indicator concentrations are again associated with sites in the River Calder, Darwen and Douglas catchments, probably related to sewerage system overflows which discharge during high flow conditions in the urbanised areas of these catchments. Spatial patterns of variation in water quality in relation to land cover in the catchment are explored further in Kay et al. (2005a).

3.3. Faecal indicator organism concentrations in treated and overflow sewage effluents

Faecal indicator organism concentrations in FE samples did not show any consistent pattern of increase or dilution in response to event conditions (Table 2). During high flow events, statistically significant elevations in GM concentrations of all three indicators were observed only at Colne WwTW with elevations of FC and EN at Wigan WwTW, for FC at Blackburn WwTW and EN at Hyndburn WwTW. In contrast, Clitheroe WwTW displayed statistically significant dilution in GM TC concentrations during high flow conditions. The worst quality of effluent for both TC and FC during base flow and high flow conditions was at Skelmersdale A WwTW FE (site 312), where high flow TC and FC GM concentrations exceeded 1.0×10^7 cfu 100 ml^{-1} (Table 2). It is interesting to note the difference in the effluent quality of the two treatment streams at Skelmersdale WwTW (Table 2).

With the exception of Horwich WwTW inlet CSO (site 333), where only one sample was obtained, at least three samples were collected from the WwTW STOs and inlet CSOs, with a maximum of seven samples being collected from Barnoldswick WwTW STO (site 324) and Croston WwTW STO (site 328) (Table 3). Maximum FC concentrations were observed at Skelmersdale WwTW STO (site 314).

Difficult access and time constraints during rainfall event conditions meant that it was only possible to collect CSO effluent samples from Rishton Tanks CSO (site 334). The GM FC concentration calculated from the four samples collected from this site is shown in Table 3. Storm sewage overflow effluent concentrations in effluents discharged

Table 2
Geometric mean and standard deviation of $\log_{10}$ transformed faecal coliform concentrations (cfu 100 ml^{-1}) at treated effluent sampling points in the Ribble catchment between 30/7/02 and 10/9/02

Site	Base flow			High flow		
	Geometric mean (cfu 100 ml^{-1})	S.D.[a]	n[b]	Geometric mean (cfu 100 ml^{-1})	S.D.[a]	n[b]
301 Wigan WwTW	6.29×10^5	0.546	13	3.88×10^{6c}	0.603	7
303 Hyndburn WwTW	4.34×10^5	0.474	12	1.09×10^6	0.802	8
305 Blackburn WwTW	7.27×10^5	0.614	13	3.33×10^{6c}	0.650	6
308 Burnley WwTW	4.04×10^4	0.755	11	8.57×10^4	0.746	9
310 Darwen WwTW	2.15×10^5	0.272	13	2.90×10^5	0.644	6
312 Skelmersdale A WwTW	5.05×10^6	0.917	14	2.09×10^7	1.118	5
313 Skelmersdale B WwTW	1.25×10^6	0.815	14	5.49×10^6	1.118	5
315 Chorley WwTW	2.11×10^5	0.567	11	1.33×10^5	0.838	8
317 Walton-le-Dale WwTW	1.70×10^5	0.546	13	6.40×10^5	0.494	4
319 Horwich WwTW	5.11×10^4	0.772	14	1.63×10^5	1.067	5
321 Leyland WwTW	4.96×10^4	0.774	14	7.11×10^4	0.845	5
323 Barnoldswick WwTW	2.57×10^5	0.354	11	2.83×10^5	0.253	8
325 Colne WwTW	1.67×10^5	0.479	14	4.96×10^{5c}	0.233	6
327 Croston WwTW	6.89×10^5	0.444	18	–	–	0
329 Clitheroe WwTW	1.29×10^6	0.227	13	6.20×10^5	0.502	7
331 Settle WwTW	2.93×10^4	1.173	16	2.82×10^4	0.577	4

[a] S.D. = standard deviation of $\log_{10}$ transformed concentrations.
[b] n = number of observations.
[c] Results of Student's t-test show a significant difference in geometric mean at high flow compared to base flow at $\alpha = 0.05$ (95% confidence).

Table 3
Geometric mean and standard deviation of $\log_{10}$ transformed faecal coliform concentrations (cfu 100 ml^{-1}) at overflow sampling points in the Ribble catchment between 30/7/02 and 10/9/02

Site	Geometric mean (cfu 100 ml^{-1})	S.D.[a]	n[b]
302 Wigan STO	9.73×10^6	0.516	5
304 Hyndburn STO	6.81×10^5	0.529	5
306 Blackburn STO	6.44×10^6	0.563	5
307 Blackburn inlet CSO	2.77×10^6	0.652	3
309 Burnley STO	4.31×10^6	0.334	4
311 Darwen STO	1.75×10^6	0.651	3
314 Skelmersdale STO	1.69×10^8	0.653	3
316 Chorley STO	6.35×10^6	0.531	6
318 Walton-le-Dale STO	4.58×10^6	0.354	4
320 Horwich STO	9.79×10^6	0.380	4
322 Leyland STO	1.80×10^6	0.272	4
324 Barnoldswick STO	5.21×10^5	0.331	7
326 Colne STO	4.70×10^6	0.175	3
328 Croston STO	7.10×10^6	0.401	7
330 Clitheroe STO	5.81×10^6	0.422	4
332 Settle STO	7.91×10^5	1.191	4
333 Horwich inlet CSO	5.33×10^6	–	1
334 Riston Tanks	1.31×10^6	0.429	4
All overflow samples	3.54×10^6	0.714	76

[a] S.D. = standard deviation of $\log_{10}$ transformed concentrations.
[b] n = number of observations.

from unsampled CSOs were characterised by the GM of all effluent overflow samples (i.e. GM of all WwTW storm tank overflow samples and the four samples from Rishton Tanks CSO) (Table 3).

3.4. *Faecal indicator organism budget for the Ribble Estuary*

The faecal indicator organism budgets for inputs to the Ribble Estuary displayed results consistent with the desk study described in Wither et al. (2005). Diffuse catchment sources (i.e. the rivers) accounted for 99% of the discharge (i.e. water quantity) budget, 57% of this being discharged during high flow events (Table 4). The River Ribble contributed the largest proportion of the flow (74%), the Rivers Darwen and Douglas contributing 8%–9% of the flow each and the Rivers Yarrow and Lostock 3%–4% each (Table 4).

An estimated 5.24×10^{17} FC were discharged to the Ribble Estuary over the period 30/7/04 to 10/9/04, of which rivers receiving upstream sewage effluent discharges delivered 90% during high flow events (Table 4). The principal riverine input was the River Douglas (site 701), which accounted for over 62% of the FC load during both base flow and high flow conditions, despite accounting for only 8% of the discharge volume (Fig. 3). The second largest riverine input to the estuary was the River Darwen (site 201), contributing 18% of the total FC load to the estuary (Table 4). The River Ribble at Samlesbury (site 101) was estimated to represent 8% of the total FC load to the estuary, despite accounting for three quarters of the flow volume (Table 4). The Rivers Lostock and Yarrow each contributed less than 3% of the total bacterial input to the Ribble Estuary (Table 4).

The hourly time series of rainfall (mm), FC flux (organisms s^{-1}) and proportional contributions (%) show

Table 4
Percentage contribution of discharge and faecal coliform inputs to the Ribble Estuary between 30/7/02 and 10/9/02

	% Contribution to discharge budget			Faecal coliform load (%)		
	Base flow	High flow	Total flow	Base flow	High flow	Total flow
101 R. Ribble	31.66	42.59	74.24	0.64	7.68	8.32
201 R. Darwen	4.16	4.98	9.14	0.87	16.70	17.57
501 R. Lostock	1.51	2.62	4.12	0.14	2.83	2.97
601 R. Yarrow	1.62	2.15	3.77	0.15	2.32	2.48
701 R. Douglas	3.54	4.62	8.16	6.89	59.99	66.87
317 Walton WwTW FE	0.28	0.16	0.44	<0.01	0.38	0.39
328 Croston WwTW FE	0.05	0.02	0.07	0.12	<0.01	0.12
318 Walton WwTW STO	–	0.04	0.04	–	0.68	0.68
329 Croston WwTW STO	–	0.02	0.02	–	0.61	0.61
Total (rivers)	42.48	56.95	99.43	8.69	89.52	98.20
Total (WwTW FE)	0.33	0.18	0.51	0.12	0.39	0.51
Total (WwTW STO)	–	0.06	0.06	–	1.29	1.29
Total (all sources)	42.81	57.19	100.00	8.81	91.19	100.00

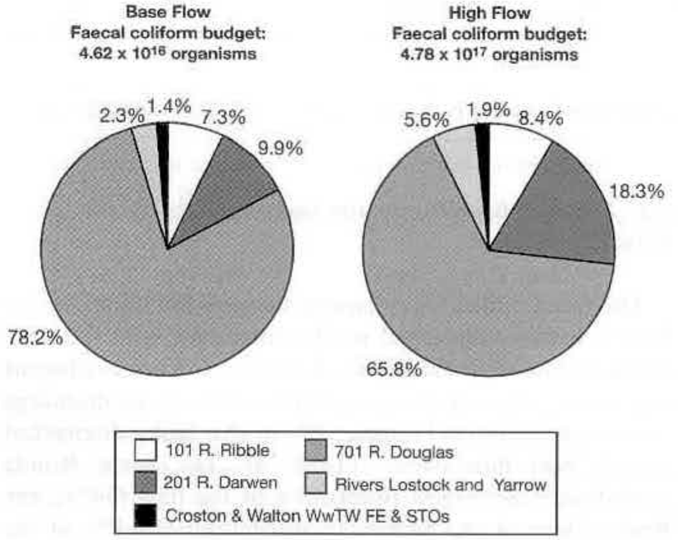


Fig. 3. Estimated base flow and high flow faecal coliform budgets for the Ribble Estuary.

that delivery of faecal indicator organisms increased considerably in response to rainfall events (Fig. 4). During base flow conditions, the River Douglas clearly dominated the inputs, contributing between 75% and 85% and peaking at over 95% of the delivery. The proportion attributable to this source decreased during high flow events, at times to less than 30% of the total delivery. During the high flow events, the proportional contribution of the River Ribble represented between 15% and 60% of the load input into the estuary. During base flow conditions this river generally accounted for less than 10% of the load. The River Darwen accounted for about 10%–20% of the load during base flow conditions, this proportion increasing to between 20% and 30% during high flow events, although during one event the contribution from this source exceeded 50%. During high flow conditions the contribution from WwTWs, including inputs from the STO sources increased to up to 15%. It should be noted that the high delivery of organisms during these times means that these proportions can represent a significant number of organisms.

3.5. *Faecal indicator organism budget for the River Douglas*

Of the $1.50 \times 10^7\,\text{m}^3$ total volume discharged at the tidal limit of the River Douglas, 56% was discharged during base flow conditions (Table 5). The River Douglas contributed the largest flow volume (43%), 24% of which was discharged during base flow conditions. Treated sewage effluent accounted for 35% of the flow volume, 25% of which was discharged during base flow conditions (Table 5). A further 2.8% of the flow was attributable to STOs.

An estimated total of 2.13×10^{17} FC were discharged at the tidal limit, 77% of which was discharged under high flow conditions (Table 5). The primary source of faecal indicator organisms in the catchment was the combined outfall of Wigan and Skelmersdale WwTWs, which accounted for between 98% of the base flow FC load and 92% of the high flow FC budget to the tidal limit (Fig. 5). Final effluents from the Skelmersdale A and Wigan WwTWs represented the major contributors to both base flow and high flow budgets. The STO at Wigan WwTW contributed an additional 24% of the high flow FC budget, despite accounting for <0.1% of the flow volume and operating for 69 h of the 1056 h of the study (i.e. 6.53% of the time). Again, the temporal plots of rainfall, organism flux and proportional contribution of each source indicate that increased faecal indicator organism delivery was related to rainfall with peak loads associated with the discharge from Wigan WwTW STO (site 302) (Fig. 6). When the STO at Wigan WwTW was not operating, the delivery of organisms was dominated by FE sources with

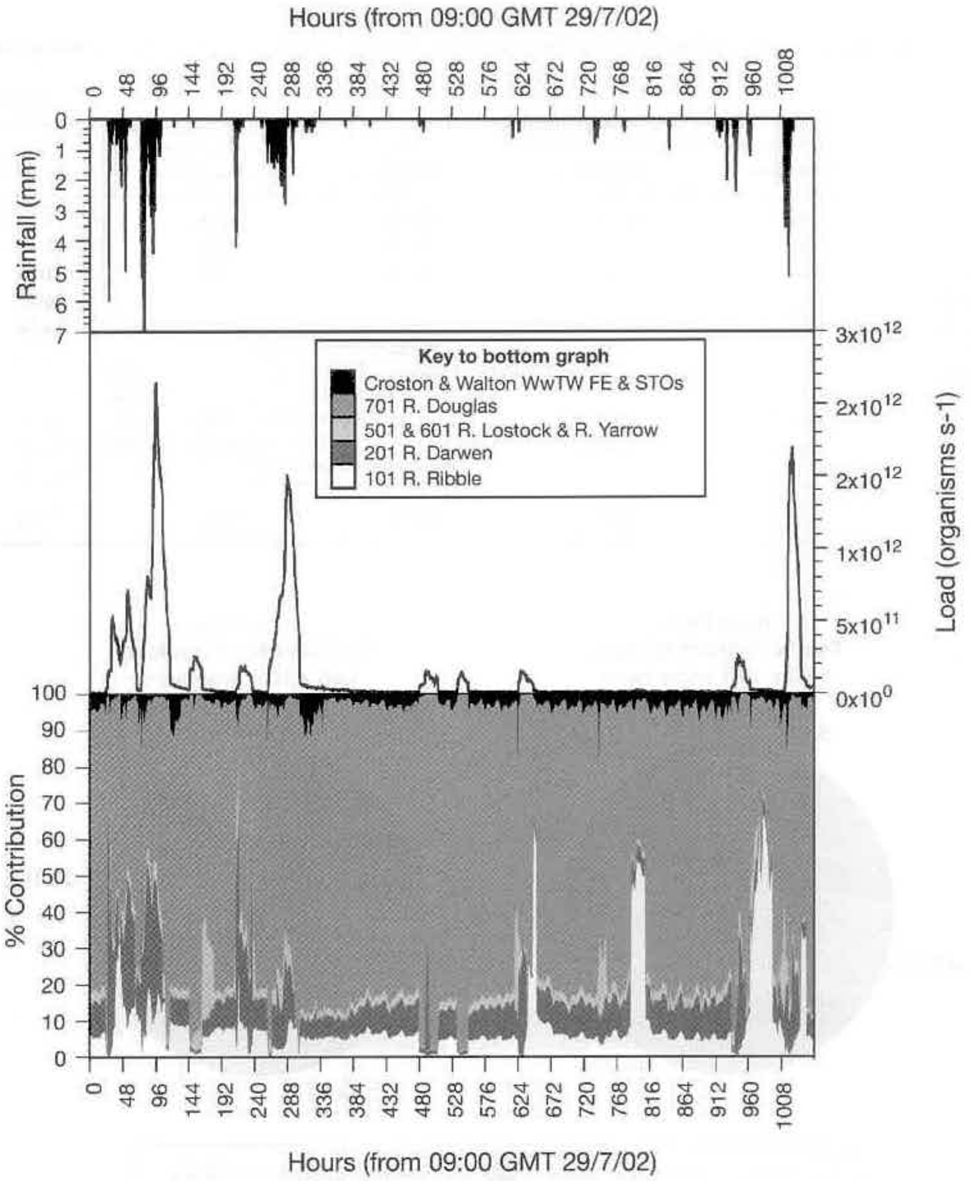


Fig. 4. Hourly rainfall (mm) at Moor Park, Preston, instantaneous faecal coliform load (organisms s^{-1}) and hourly proportional contributions (%) to the hourly load input to the Ribble Estuary tidal limits.

riverine sources contributing less than 20% of the total during high flow events.

Upstream of the Wigan/Skelmersdale WwTW outfall, water quality in the River Douglas was much better, with estimated loads at Martland Bridge (site 702) less than 10% of those at Wanes Blades Bridge. The loads from the River Tawd were less than 1% of the FC load estimate for the river at Wanes Blades Bridge.

Since the field data collection for this study was completed, ultra-violet (UV) disinfection has been added to the process stream at both Wigan and Skelmersdale WwTWs. In the absence of empirical data for these effluents at the time of the study, the potential impact of the UV disinfection on bacterial loads was estimated by applying a $2\log_{10}$ reduction to the base flow and high flow GM concentrations at Wigan and Skelmersdale 'A' and 'B' WwTWs. This reduction was considered to be an achievable performance of the UV plant. Such a reduction in FC concentrations would reduce the estimated FC budget by 74%. During base flows, FE from Wigan and Skelmersdale WwTWs would account for 33% of the FC load of the River Douglas (1.4×10^{15} organisms) (compared to 98% of 4.8×10^{16} organisms pre installation UV treatment) and 0.2% of the Ribble Estuary budget (12.4% pre-UV). During high flow events, the proportional load from the FEs would decrease from 68% of 1.6×10^{17} organisms to 2% of 5.5×10^{16} organisms. The contribution to the estuary budget would decrease from 43% pre-UV to 18% following UV treatment. The total delivery of organisms to the River Douglas would be dominated by STO sources, accounting for 72% of the FC budget. Less than 3% of the total load of FC would be delivered during base flow conditions, when riverine inputs from the Rivers Douglas (site 702) and Tawd (site 703) would dominate.

Table 5
Percentage contribution of discharge and faecal coliform inputs to the to the River Douglas upstream of Wanes Blades Bridge (site 701) between 30/7/02 and 10/9/02

	% Contribution to discharge budget			Faecal coliform load (%)		
	Base flow	High flow	Total flow	Base flow	High flow	Total flow
702 R. Douglas	23.54	19.65	43.19	0.28	4.87	5.15
703 R. Tawd	7.01	12.18	19.20	0.16	1.38	1.54
301 Wigan WwTW FE	21.04	7.47	28.50	9.98	21.66	31.64
312 Sk'dale A WwTW FE	2.64	1.58	4.22	10.82	26.87	37.70
313 Sk'dale B WwTW FE	1.33	0.80	2.13	1.32	3.74	5.05
302 Wigan WwTW STO	–	2.73	2.73	–	18.74	18.74
314 Sk'dale WwTW STO	–	0.03	0.03	–	0.17	0.17
Total (rivers)	30.55	31.84	62.38	0.45	6.25	6.69
Total (WwTW FE)	25.00	9.85	34.85	22.12	52.27	74.39
Total (WwTW STO)	–	2.76	2.76	–	18.92	18.92
Total (All Sources)	55.55	44.45	100.00	22.56	77.44	100.00

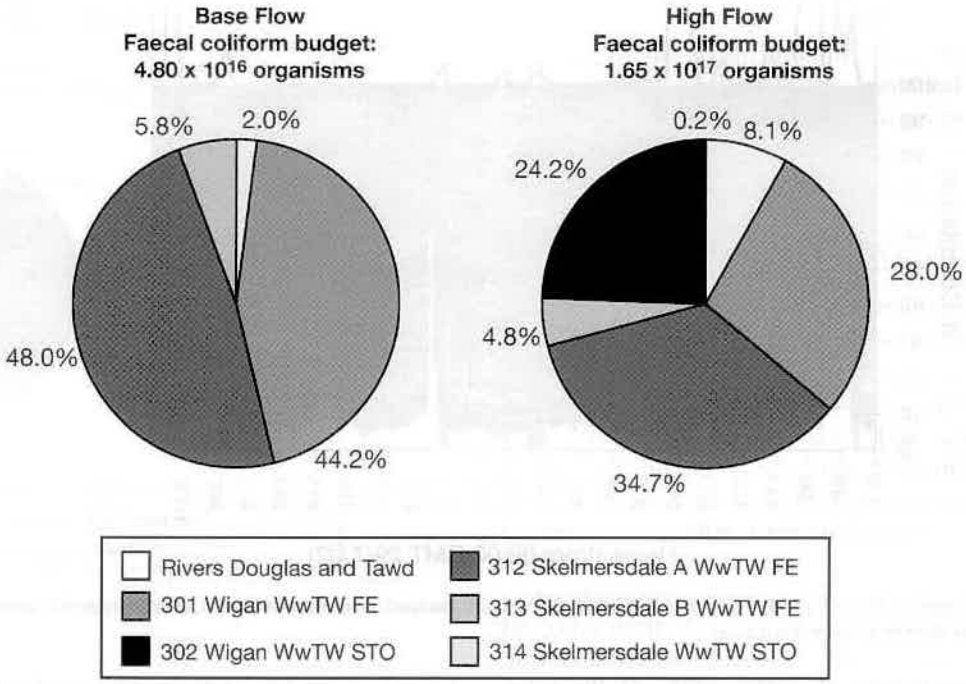


Fig. 5. Estimated base flow and high flow faecal coliform budgets for the River Douglas upstream of the tidal limit.

During high flow events, inputs from Wigan WwTW STO (site 302) would account for over 70% of the high flow FC budget. This illustrates the need to consider the contribution of intermittent storm overflow discharges during the design of schemes to reduce faecal inputs from sewage discharges.

3.6. Other subcatchment faecal indicator organism budgets

Other subcatchment budget estimates were also constructed for contributions to the tidal limit of the River Darwen, for the confluence of the Rivers Ribble, Calder and Hodder, and for the River Calder upstream of its confluence with the River Ribble.

The River Darwen catchment represented the second largest input of faecal indicator organisms to the Ribble Estuary. High flow FE and network CSO sources contributed the majority (96%) of the bacterial load in approximately equal proportions. The major source of FE in the catchment, Blackburn WwTW, dominated base flow budgets whilst, sources upstream of the Blackburn WwTW discharge, which included Darwen WwTW, contributed a similar high flow load to Blackburn WwTW FE.

Of the three main subcatchments in the River Ribble catchment (Rivers Hodder, Calder and mid/upper Ribble), the River Calder was estimated to contribute over half of the faecal indicator organism loads. Upstream of the confluence in the Calder subcatchment, the base flow

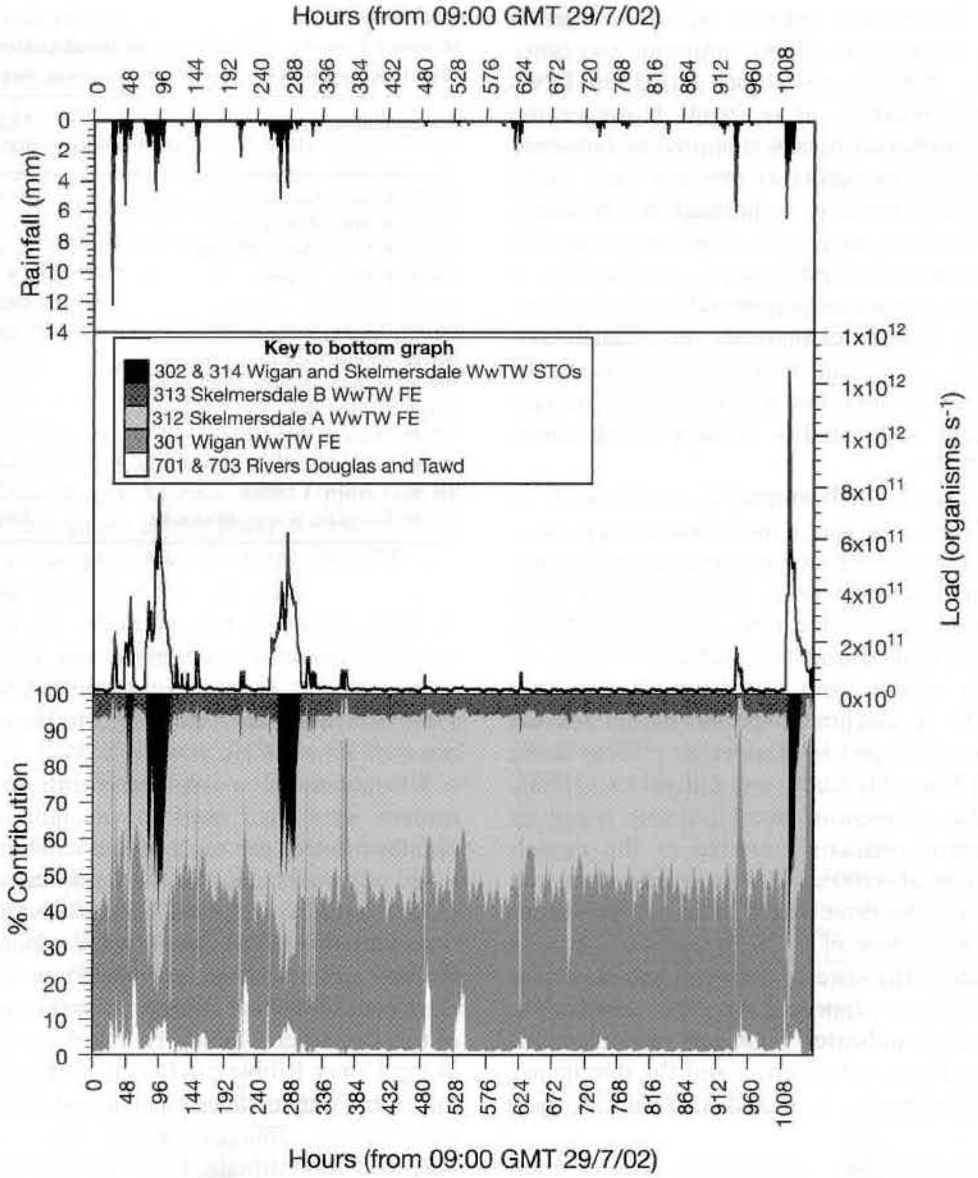


Fig. 6. Hourly rainfall (mm) at Common Bank, instantaneous faecal coliform load (organisms s^{-1}) and hourly proportional contributions (%) to the hourly load input to the River Douglas upstream of Wanes Blades Bridge (site 701).

bacterial budgets are dominated by Hyndburn WwTW FE. During high flow conditions, the River Calder itself contributes the majority of organisms, although this includes treated and storm inputs from three WwTWs. Bacterial loads in the Ribble catchment upstream of the confluence with the River Calder are dominated by non-sewage related diffuse sources. Estimated loads from the small WwTWs and network CSOs within the Hodder catchment show their contributions to be relatively insignificant when compared to catchment diffuse sources.

4. Discussion: assessing the contribution of catchment diffuse sources

The various faecal indicator organism budgets described above provide an indication of the potential sources of microbial contamination at key points in the catchment. However, the riverine components comprise all inputs upstream of the monitoring point whether they are from continuous or intermittent point sources or catchment diffuse sources and this latter element is particularly difficult to quantify in isolation. Diffuse sources can include direct defecation from livestock into watercourses, runoff of faecal matter from grazing areas and/or areas receiving manure applications during periods of rainfall and contamination from runoff from farmstead areas. Clearly, even if it were possible to monitor all such sources, the size of the Ribble catchment would preclude such a strategy due to high number of monitoring points required.

Further to elucidate the contribution of catchment diffuse sources, it is necessary to employ other methods. Such methods include chemical or microbial tracer studies (e.g. Rhodamine, *Bacillus globigii*) and chemical, biochemical and molecular faecal pollution source tracking techniques (e.g. fluorescent optical whitening agents (Hagedorn, 2001; Poiger et al., 1996), antibiotic resistance analysis (Wiggins, 1996) or techniques based on DNA markers or genotyping (Meays et al., 2004). However, the use of chemical or microbial tracers is limited to following only a limited number of inputs at any one time whilst faecal pollution source tracking techniques are relatively new, no standard methods have yet been proposed, sample analysis is often complicated and requires costly analytical equipment and many techniques require reference libraries. The application of these latter methods on a catchment-wide basis is still uncertain and more comparison studies are required in order to determine which methods are best suited to catchment scale studies (Meays et al., 2004; Stapleton et al., 2007).

An alternative approach to empirical studies such as those outlined above is to use a model-based approach. Statistical models relating digital land cover data to water quality have been developed using empirical data from similar catchment studies which can be applied both to the specific catchments studied and to catchments where no data exists using a generic form of the model (Crowther et al., 2002, 2003). A catchment-specific model for the Ribble catchment, developed by Kay et al. (2005a) using the empirical data from this study, was utilised to estimate the catchment diffuse element of faecal indicator organism input. The dominant predictor variable in the models developed by Kay et al. (2005a), for both base flow and high flow models for the three faecal indicator organisms studied, was the proportion of built-up land. This can be considered an index of the sewered area and hence, sewage inputs, an assumption supported by the increase in geometric mean faecal indicator organism concentrations downstream of the major urban areas and the dominance of the sewage related inputs associated with each of these areas.

In order to estimate the agricultural diffuse input element excluding point-source sewage inputs, GM faecal indicator concentrations were predicted for a hypothetical situation assuming a zero proportion of urban area, the predicted water quality being based on the proportions of remaining land cover types (e.g. improved pasture) in the subcatchments. The predicted high flow geometric mean faecal indicator organism concentrations (when the majority of the organisms are delivered) for the diffuse only sources at the outlets of the main subcatchments are presented in Table 6. The predicted high flow geometric mean concentrations were noticeably lower than observed at the outlets of catchments particularly impacted by sewage discharges (e.g. sites 201 (Darwen), 501 (Lostock) 601 (Yarrow) and 701 (Douglas). Indeed, the predicted high flow FC geometric mean concentration at the tidal limit of the River Douglas (site 701) (5.8×10^3 cfu $100\,\text{ml}^{-1}$) was almost three orders of magnitude lower than the observed concentration (3.6×10^6 cfu $100\,\text{ml}^{-1}$). In contrast, subcatchments with few sewage inputs displayed predicted geometric mean concentrations similar to those observed. For example, the predicted high flow geometric mean FC concentration for the upper Ribble catchment at Horton in Ribblesdale (site 107) was 1.6×10^4 cfu $100\,\text{ml}^{-1}$ compared to the observed geometric mean of 1.7×10^4 cfu $100\,\text{ml}^{-1}$.

Table 6
Modelled high flow geometric mean faecal coliform concentrations (cfu $100\,\text{ml}^{-1}$) at subcatchment outflows assuming zero sewage inputs

Subcatchment	Faecal coliforms (cfu $100\,\text{ml}^{-1}$)
101 R. Ribble, Samlesbury	3.78×10^4
102 R. Ribble, Ribchester	3.65×10^4
103 R. Ribble, Great Mitton	6.99×10^4
104 R. Ribble, Sawley	6.83×10^4
105 R. Ribble, Cow Bridge	3.93×10^4
106 R. Ribble, Settle Weir	2.25×10^4
107 R. Ribble, Horton-in-R'dale	1.57×10^4
109 R. Hodder, Lower Hodder Br.	2.05×10^4
120 R. Calder, Whalley Weir	2.00×10^4
201 R. Darwen, Blue Bridge	7.94×10^3
501 R. Lostock	4.64×10^4
601 R. Yarrow, Croston	7.12×10^3
701 R. Douglas, Wanes Blades Br.	5.80×10^3

The geometric mean concentrations for diffuse only sources were then used to calculate the corresponding organism loads for each of the catchment outlets. These loads were compared to the loads calculated for FE and STO sources from WwTWs and overflow contributions from network CSOs. No WwTW inputs within the River Hodder catchment were included in the study although there are some small treatment works and associated CSOs in this catchment. Similarly, there are some sewage inputs in the Upper Ribble catchment upstream of Settle WwTW that were not included. To provide an estimate of loads from sewage sources within these catchments, it was necessary to estimate loads from these small WwTWs. FE loads were estimated using their consented dry weather flow (DWF) and maximum flow scaled using the high flow profile for Settle WwTW (selected because it is the smallest monitored WwTW). STO loads were estimated by applying the proportion of total flow from the 15 WwTWs discharged from STOs (6.96%) whilst CSO inputs were calculated using the proportion of total flow from the 15 WwTWs estimated to be discharged from CSOs (28.79%). It was felt necessary to include estimates for these sources in these catchments due to the potentially low loads from diffuse sources and to enable a more appropriate comparison with the other subcatchments.

The resultant high flow FC loads from catchment diffuse, WwTW FE, WwTW STOs and network CSOs are shown in Fig. 7 whilst Table 7 shows proportional contributions of each source type to the estimated load for FC in each subcatchment. Data for the River Douglas catchment are presented for both the Summer 2002 situation (i.e. secondary treatment of Wigan and Skelmersdale WwTW FEs) and after planned UV disinfection.

Fig. 7 clearly illustrates the difference in high flow faecal indicator organism loads from the various subcatchments. Relatively low loads were derived from the upper Ribble and Hodder catchments whilst the largest loads were derived from the highly urbanised catchments of the Rivers Calder, Darwen and Douglas. The lower Ribble also contributed a relatively high load, second only to the pre-UV load from the River Douglas although this includes the contribution of the Rivers Calder and Hodder as well as sources along its own course.

The estimated high flow FC fluxes in the upper Ribble and Hodder catchments are dominated by diffuse catchment sources, accounting for over 98% of the load, reflecting the low degree of sewerage infrastructure within these catchments (Table 7). Catchment diffuse sources also dominate in the combined Middle and Upper Ribble catchment (i.e. the catchment of the River Ribble to Great Mitton), although the presence of Settle WwTW and associated sewerage infrastructure decreases the proportional contribution of diffuse catchment sources to 68% whilst the majority of the remaining load is associated with CSO inputs (Table 7). However, these catchments account for a small proportion of the total estimated load contributed to the Ribble Estuary (Upper Ribble and Hodder catchments <1%; Middle Ribble catchment <10%).

WwTW FE contributes the largest proportion of high flow FC subcatchment loads in the River Douglas (56%) and River Darwen (43%) catchments (Table 7), which contain the largest WwTWs in the study. The improvements implemented to Wigan and Skelmersdale WwTWs were estimated to decrease the FE contribution in the Douglas catchment to less than 2% (Table 7). Relatively high proportions of subcatchment loads associated with WwTW STOs are present in the Yarrow (42%) and post-improvement Douglas (56%) catchments. In the remaining subcatchments (i.e. Calder, Lower Ribble and Lostock, as well as in the Yarrow catchment), CSO inputs account for the largest proportion of the estimated high flow FC load (Table 7; Fig. 7).

The Douglas catchment accounts for the greatest proportion of the estimated load to the Ribble Estuary (41%) although the improvements to sewage treatment

Table 7
Percentage of high flow faecal coliform load from each source category

Subcatchment	Catchment diffuse	Final effluent	Storm tank overflows	Network CSOs
Upper Ribble	99.54	0.28	0.18	–
Mid Ribble	68.29	3.49	3.83	24.39
Lower Ribble	22.32	10.92	12.65	54.10
Calder	4.14	14.13	16.46	65.28
Hodder	98.22	0.39	–	1.39
Darwen	0.59	42.74	1.62	55.05
Douglas[a]	0.19	55.76	25.04	19.01
Douglas UV[b]	0.42	1.72	55.63	42.23
Yarrow	1.14	2.08	41.83	54.95
Lostock	13.45	0.76	8.55	77.24
Estuary[a]	6.76	35.60	17.00	40.63
Estuary UV[b]	8.73	16.80	21.97	52.50

The catchment diffuse element was estimated using predicted geometric mean faecal coliform concentrations shown in Table 6.

[a] Estimated loads during field study phase (summer 2002).

[b] Estimated loads assuming ultra-violet disinfection of FEs from Wigan and Skelmersdale A and B WwTWs.

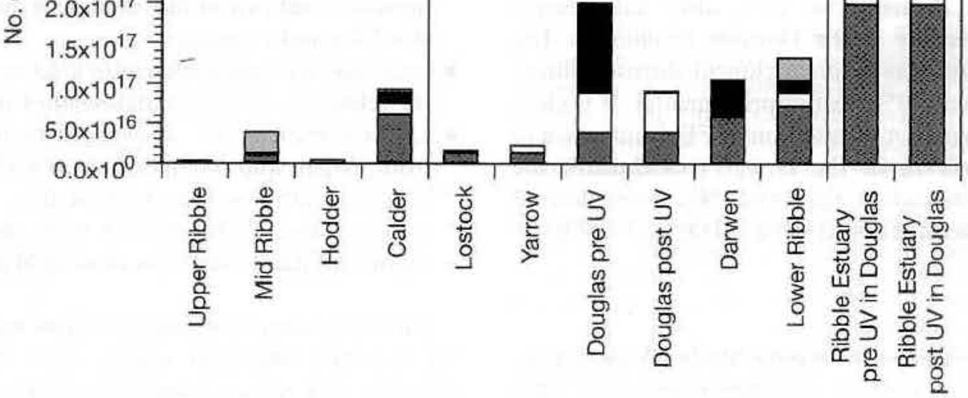


Fig. 7. Estimated high flow faecal coliform loads discharged to the Ribble Estuary and selected subcatchments from diffuse catchment sources and sewage effluents.

Table 8
Percentage of high flow faecal coliform load from each subcatchment contributing to the load input to the Ribble Estuary estimated using the catchment outflow load and modelled/estimated load by source (pre- and post-UV treatment at Wigan and Skelmersdale WwTWs)

Subcatchment	Catchment outflow load[a]	Estimated load by source[b]	
		Pre AMP3[c]	Post AMP3[d]
Upper Ribble	<0.01	0.61	0.79
Mid Ribble	6.44	8.12	10.49
Lower Ribble	8.42	27.12	35.04
Calder	10.20	18.96	24.50
Hodder	3.08	0.67	0.86
Darwen	18.31	21.54	27.83
Douglas	65.78	41.11	23.91
Yarrow	2.55	4.56	5.89
Lostock	3.10	3.34	4.32
Direct to estuary	1.83	2.33	3.01

[a] Calculated using measured water quality at the catchment outflow.

[b] Calculated by adding estimated loads from sewage sources (FE+STOs+CSOs) to a modelled catchment load assuming zero built-up area (i.e. an estimate of diffuse catchment sources excluding sewage sources).

[c] Situation during field study phase – secondary treatment of FE and Wigan and Skelmersdale A and B WwTWs.

[d] Scenario assuming UV treatment with $2\log_{10}$ reduction of secondary treated FE bacterial concentrations at Wigan and Skelmersdale A and B WwTWs.

implemented since the data collection exercise decrease the contribution of this catchment to 24% of the reduced total load to the estuary (Table 8). Following these improvements, the Lower Ribble catchment was estimated to contribute the largest proportion of the total load to the estuary (35%), although this predominantly comprises the loads from Middle Ribble and Calder catchments. Of these three catchments, the Calder catchment dominates the post-improvement load prediction accounting for 25% of the bacterial contribution. The other large contributor to the estuary loads was the Darwen catchment, accounting for 28% of the post-improvement FC load (Table 8).

Of the estimated load to the Ribble Estuary (i.e. from the whole catchment) the largest proportion of FC was from CSO sources (41%) followed by FE (36%) (Table 7). This was estimated to decrease to 17% after the sewage treatment improvements in the Douglas catchment. The smallest contribution was from catchment diffuse sources (7% pre-improvement, 9% post-improvements). It is clear from Fig. 7 that even after installation of FE disinfection to the two large WwTWs in the Douglas catchment, the majority of faecal indicator organisms (74%) were derived from intermittent sewage sources (i.e. STOs and CSOs).

5. Conclusions

The desk study phase of the Ribble Study (Wither et al., 2005) illustrated the type of catchment 'profiling' that might be undertaken to implement the requirements of the WFD or health-related standards for bathing waters recommended by CEC (2006), WHO (1999, 2003). However, faecal indicator organism data are rarely available from historical routine monitoring and moderate additional sampling effort would be required even to undertake such a desk exercise.

The desk study also made many assumptions and if crucial expenditure decisions and/or legally required environmental compliance hinged on the outcome it would be sensible to confirm these assumptions via additional empirical monitoring. For these reasons, a second phase of empirical data acquisition from rivers and point sources of faecal indicators was undertaken in the summer 2002 bathing season. The resultant budgets constructed from the Phase II study data displayed results consistent with the desk study described in Wither et al. (2005). The good agreement of budgets suggests that the assumptions made during the desk study were broadly appropriate.

Faecal indicator concentrations in rivers and streams draining to the Ribble Estuary showed significant elevations, by an order of magnitude or more, during hydrograph response to rainfall. This pattern was similar to that observed in previous catchment studies. An analysis of discharge in the River Ribble for the same 44-day period as the study between 1991 and 2002 suggested that environmental conditions during the study were somewhat atypical compared to the long term record containing some extreme events which may have produced higher faecal indicator concentrations. Again, however, the good agreement of the results described herein with the earlier studies (Fewtrell et al., 1998; Wither et al., 2005) suggests that these extreme events did not have a disproportionate effect on the budget estimates.

Faecal indicator organism budgets for inputs to the Ribble Estuary and within the catchment of the River Ribble showed the following broad patterns during the study period:

- the flux of organisms to the estuary was dominated by sewage related sources discharged during high flow episodes, accounting for over 90% of the total organism load;
- sewage sources are largely related to the urban areas in the south and east of the catchment with their associated WwTWs and overflows;
- over half the faecal indicator load was associated with the relatively small subcatchment of the River Douglas;
- UV treatment of FE discharged to the River Douglas from Wigan and Skelmersdale WwTWs was estimated to considerably reduce the base flow load from sewage sources although during high flows, loads would still be significant due to the operation of Wigan WwTW STO.

The most surprising finding of this empirical study was the fact that untreated sewage spills from urban infrastructure and sewage treatment works are the dominant component of the high flow flux of faecal indicators to the Ribble Estuary and receiving waters of the Fylde coast.

Indeed, the volume of storm sewage discharged during the study period from the overflow at Wigan WwTW was greater than the volume of treated sewage discharged by four of the fifteen WwTWs (i.e. Barnoldswick, Colne, Croston and Settle WwTWs), the largest of which has a population equivalent of over 26,000. It is therefore clear that future capital investment on sewerage infrastructure within the Ribble catchment should address the issue of storm sewage discharges, which, together with sustainable reduction in diffuse source pollution, would be needed to effect further improvement in coastal water quality (Kay et al., 2005b, c, 2007a, b).

This study of the faecal indicator organism budgets in the Ribble catchment differs from previous similar studies (Crowther et al., 2002, 2003; Kay et al. 2005b; Wyer et al., 1996, 1998) in that the Ribble catchment contains a much greater proportion of urbanised land, consequently containing a larger number of WwTWs and storm sewage overflows. This may, in part explain the dominance of the sewage-related inputs within this particular catchment. The other studies noted above were undertaken in predominantly rural areas with a low proportion of built-up land cover, and diffuse catchment sources of faecal indicator organisms were dominant. Nevertheless, this study does highlight the necessity of including intermittent storm sewage overflow inputs when deriving catchment faecal indicator organism budgets.

Acknowledgements

The study was funded by the Environment Agency, North West Region. The views expressed in this paper are those of the authors and do not necessarily reflect those of the Environment Agency. The authors wish to thank the following people who contributed to the study: Gary Beasley, Andrew Davies, Matthew Hopkins, Nikki Kay, Colin Last, Nicola Poulter, Sarah Stapleton and Lee Whitehead (CREH) – sample collection and transport; Paula Hopkins (CREH) – field logistics organization; Michael Shankster, Robb Turner, Gareth Owen, Wally Ball, Phil Heath, Terry Bucknall, Carol Holt, Andy Brown and Helen Rice (Environment Agency) – GIS, hydrometry and field logistical support; Hanna Boynton, Diane Corscadden and Lenka Rushby (CREH Analytical Ltd) – laboratory analytical assistance.

References

Anon. 2002. Federal Water Pollution Control Act (As amended through P.L. 107-303, November 27 2002). US Environmental Protection Agency, Washington DC ⟨http://www.epa.gov/r5water/cwa.htm⟩.

Anon. 2005. Total Maximum Daily Loads. US Environmental Protection Agency, Washington DC ⟨http://www.oaspub.epa.gov/waters/national_report.control⟩.

CEC. 1976. Council Directive 76/160/EEC of 8 December 1975 concerning the quality of bathing water. Official Journal of the European Communities L31, 1–7.

CEC, 2000. Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy. Official Journal of the European Communities L327, 1–72.

CEC, 2006. Directive 2006/7/EC of The European Parliament and of The Council of 15th February 2006 concerning the management of bathing water quality and repealing Directive 76/160/EEC. Official Journal of the European Union L64, 37–51, (4.3.2006).

Crowther, J., Kay, D., Wyer, M., 2002. Faecal-indicator concentrations in waters draining lowland pastoral catchments in the UK: relationships with land use and farming practice. Water Research 36, 1725–1734.

Crowther, J., Wyer, M.D., Bradford, M., Kay, D., Francis, C.A., 2003. Modelling faecal indicator concentrations in large rural catchments using land use and topographic data. Journal of Applied Microbiology 94, 962–973.

Defra, 2003. Strategic Review of Diffuse Water Pollution from Agriculture: Discussion Document. Department for Environment, Food and Rural Affairs, London.

Environment Agency, 2000. The Microbiology of Recreational and Environmental Waters 2000. Methods for the Enumeration of Waters and Associated Material. Standing Committee of Analysts. The Environment Agency, Bristol, 84 pp.

Fewtrell, L., Kay, D., Wyer, M., Crowther, J., Carbo, P., Mitchell, G., 1998. Faecal indicator budgets discharging to the Ribble Estuary. Report to the Environment Agency, North West Region. 52 pp.

Fleisher, J.M., McFadden, R.T., 1980. Obtaining precise estimates in coliform enumeration. Water Research 14, 477–483.

Haygarth, P., Granger, S., Chadwick, D., Sheperd, M., Fogg, P., 2005. A Provisional Inventory of Diffuse Pollution Losses. ADAS Report for Department for Environment, Food and Rural Affairs, London.

Hagedorn, C., 2001. Bacterial Source Tracking. Crop and Soil Environmental Sciences, VA Tech ⟨http://filebox.vt.edu/users/chagedor/biol_4684/BST/BSTmeth.html⟩.

Kay, D., Fleisher, J.M., Salmon, R.L., Jones, F., Wyer, M.D., Godfree, A.F., Zelenauchjacquotte, Z., Shore, R., 1994. Predicting likelihood of gastroenteritis from sea bathing – results from randomized exposure. Lancet 344 (8927), 905–909.

Kay, D., Bartram, J., Prüss, A., Ashbolt, N., Wyer, M.D., Fleisher, J.M., Fewtrell, L., Rogers, A., Rees, G., 2004. Derivation of numerical values for the World Health Organization guidelines for recreational waters. Water Research 38, 1296–1304.

Kay, D., Wyer, M., Crowther, J., Stapleton, C., Bradford, M., McDonald, A., Greaves, J., Francis, C., Watkins, J., 2005a. Predicting faecal indicator fluxes using digital land use data in the UK's sentinel Water Framework Directive catchment: the Ribble study. Water Research 39, 3967–3981.

Kay, D., Wyer, M.D., Crowther, J., Stapleton, C., Wilkinson, J., Glass, P., 2005b. Sustainable reduction in the flux of microbial compliance parameters to coastal bathing waters by a wetland ecosystem produced by a marine flood defence structure. Water Research 39, 3320–3332.

Kay, D., Wilkinson, J., Crowther, J., Reid, S., Francis, C., Kay, C., Hopkins, M., Watkins, J., Edwards, A., McDonald, A., Wyer, M., Stapleton, C., 2005c. Monitoring the effectiveness of field and steading measures to reduce diffuse pollution from agriculture to bathing waters in the Ettrick, Cessnock, Nairn and Sandyhills catchments. Report reference: ENV/7/4/04. Scottish Executive, Edinburgh. Available electronically from ⟨http://www.scotland.gov.uk/Topics/Environment/Water/15561/15068⟩

Kay, D., McDonald, A., Stapleton, C., Wyer, M., Fewtrell, L., 2006a. Europe: a challenging new framework for water quality. In: Proceedings of the Institute of Civil Engineers: Civil Engineering 159 (5), 58–64.

Kay, D., Ashbolt, N., Wyer, M.D., Fleisher, J.M., Fewtrell, L., Rogers, A., Rees, G., 2006b. Comment on 'derivation of numerical values for the World Health Organization guidelines for recreational waters': a reply. Water Research 40, 1921–1925.

Kay, D., Stapleton, C.M., Wyer, M.D., McDonald, A.T., Crowther, J., 2006c. Total Maximum Daily Loads (TMDL). The USEPA approach to managing faecal indicator fluxes to receiving waters: lessons for UK environmental regulation? In: Gairns, L., Crighton, C., Jeffrey, W.A., (Eds.), Agriculture and the Environment VI: Managing Rural Diffuse Pollution. Proceedings of the SAC/SEPA Biennial Conference, Edinburgh pp. 23–33.

Kay, D., Edwards, A.C., Ferrier, B., Francis, C., Kay, C., Rushby, L., Watkins, J., McDonald, A.T., Wyer, M., Crowther, J., Wilkinson, J., 2007a. Catchment microbial dynamics: the emergence of a research agenda. Progress in Physical Geography 31 (1), 59–76.

Kay, D., Aitken, M., Crowther, J., Dickson, I., Edwards, A.C., Francis, C., Hopkins, M., Jeffrey, W., Kay, C., McDonald, A.T., McDonald, D., Stapleton, C.M., Watkins, J., Wilkinson, J., Wyer, M., 2007b. Reducing fluxes of faecal indicator compliance parameters to bathing waters from diffuse agricultural sources, the Brighouse Bay study, Scotland. Environmental Pollution 147, 139–149.

Meays, C.L., Broersma, K., Nordin, R., Mazumder, A., 2004. Source tracking fecal bacteria in water: a critical review of current methods Journal of Environmental Management 73 (1), 71–79.

McDonald, A., Kay, D., 1981. Enteric bacterial concentrations in reservoir feeder streams: baseflow characteristics and response to hydrograph events. Water Research 15, 861–868.

Poiger, T., Field, J.A., Field, T.M., Giger, W., 1996. Occurrence of fluorescent whitening agents in sewage and river water determined by solid-phase extraction and high-performance liquid chromatography. Environmental Science and Technology 30 (7), 2220–2226.

Stapleton, C.M., Wyer, M.D., Kay, D., Crowther, J., Walters, M., Gawler, A., Hindle, T., 2007. Microbial source tracking: a forensic technique for microbial source identification? Journal of Environmental Monitoring 9, 427–439.

Wilkinson, J., Kay, D., Wyer, M.D., Jenkins, A., 2006. Processes driving the episodic flux of faecal indicator organisms in streams impacting on recreational and shellfish harvesting waters. Water Research 40 (1), 153–161.

Wither, A., Greaves, J., Dunhill, I., Wyer, M., Stapleton, C., Kay, D., Humphrey, N., Watkins, J., Francis, C., McDonald, A., Crowther, J., 2005. Estimation of diffuse and point source microbial pollution in the Ribble catchment discharging to bathing waters in the north west of England. Water Science and Technology 51 (3–4), 191–198.

WHO, 1999. Health Based Monitoring of Recreational Waters: The Feasibility of a new Approach (the "Annapolis Protocol). World Health Organisation, Geneva, Switzerland.

WHO, 2003. Coastal and fresh water. In: Guidelines for Safe Recreational-Water Environments, vol. 1. World Health Organisation, Geneva, Switzerland.

Wiggins, B.A., 1996. Discriminant analysis of antibiotic resistance patterns in fecal streptococci, a method to differentiate human and animal sources of fecal pollution in natural waters. Applied and Environmental Microbiology 62 (11), 3997–4002.

Wyer, M., Kay, D., Dawson, H.M., Jackson, G.F., Jones, F., Yeo, J., Whittle, J., 1996. Delivery of microbial Indicator organisms to coastal waters from catchment sources. Water Science and Technology 33 (2), 37–50.

Wyer, M., Kay, D., Crowther, J., Whittle, J., Spence, A., Huen, V., Wilson, C., Carbo, P.J.N., 1998. Faecal-indicator budgets for recreational coastal waters: a catchment approach. Journal of the Chartered Institution of Water and Environmental Management 12, 414–424.

ELSEVIER

Journal of Environmental Management 87 (2008) 551–559

Journal of
Environmental
Management

www.elsevier.com/locate/jenvman

Farmyards, an overlooked source for highly contaminated runoff

Anthony C. Edwards[a,*], David Kay[b], Adrian T. McDonald[c], C. Francis[d], J. Watkins[d], J.R. Wilkinson[b], M.D. Wyer[b]

[a]*Nether Backhill, Ardallie, By Peterhead, Aberdeenshire, AB42 5BQ, UK*
[b]*Centre for Research into Environment and Health, IGES, University of Wales, Aberystwyth, SY23 3DB, UK*
[c]*School of Geography, University of Leeds, Leeds LS2 9JT, UK*
[d]*CREH Analytical Ltd., 50 Back Lane, Horsforth, Leeds, West Yorkshire LS18 4RS, UK*

Received 20 February 2006; received in revised form 19 June 2006; accepted 24 June 2006

Abstract

Summer sampling of storm runoff generated from areas of roofs and hardstanding situated on four dairy/beef farms has provided novel information regarding its microbiological and chemical quality. All farm hardstandings generated runoff that was contaminated with respect to those pollutants (faecal coliforms, FC, and faecal streptococci, FS, major nutrients, organic carbon) that are ubiquitously ssociated with faecal matter and urine. The separate analysis of roof runoff indicated that these can contribute significant oncentrations of FS, phosphorus (P) and potentially toxic elements such as zinc (Zn), and suggests a level of 'background' ontamination originating from wash-off of bird droppings and in the case of Zn galvanised surfaces. On average hardstanding runoff howed enhanced concentrations of >4 orders of magnitude for FC and 2–3 for major nutrients and carbon relative to roof runoff. Organic forms of nitrogen (N) and P contributed significantly (averaging >40%) to the total dissolved fraction in both roof and hardstanding runoff. Part of the substantial variability in composition of runoff samples could be attributed to differences between farms s well as the timing of sample collection during individual storms. Where situations allowed, a comparison of water upstream and downstream of the farmyard demonstrated they acted as a source of multiple contaminants not only during hydrologically active storm vents but also during dry periods. Contamination pathways included a combination of both point (e.g., septic overflows) and non-point e.g., seepage from livestock housing) sources. Farmyards situated within intensive livestock farming areas such as SW Scotland, would be expected to have significant local and accumulated downstream impacts on the aquatic environment. Localised impacts would be particularly important for headwaters and low order streams.

Keywords: Farmyards; Hardstanding runoff; Roof runoff; Livestock; Nitrogen; Phosphorus; Potassium; Zinc; Faecal indicator organisms

---

1. Introduction

Farmyards, which include farm buildings, adjacent ivestock collecting areas, access tracks and overflows from domestic wastewater systems represent a combined environmental risk as a source for various environmental contaminants. The conditions under which these individual farm components collectively referred to here as the farmyard' represent actual contaminant sources will depend upon a range of farm management factors (e.g., length of housed period), structural design (e.g., routing of drains), geographical location (e.g., proximity to stream network) and environmental/climatic conditions (e.g., storm frequency). Although a general awareness exists regarding farmyards as potential contributors of contaminants, especially pesticides (Neumann et al., 2002) and gaseous emissions (Misselbrook et al., 2001), to surface and groundwater, few actual measurements of runoff composition or fluxes exist. Actual losses from farmyards tend not to be quantified independently and are therefore included within the more general measurement of 'catchment' losses. Recently, using an artificial rainfall generator Hively et al. (2005) demonstrated the potential for cow paths and

---

*Corresponding author. Tel.: +44 1358711244.
E-mail address: t.edwards25@btinternet.com (A.C. Edwards).

0301-4797/$ - see front matter © 2007 Published by Elsevier Ltd.
doi:10.1016/j.jenvman.2006.06.027

barnyards to generate runoff with very high concentrations of dissolved P (11.6 mg l^{-1}). In an Irish wetland study, Dunne et al. (2005) analysed soluble P concentrations between 15 and 20 mg l^{-1} in runoff from a dairy farmyard. It is reasonable to assume some fraction of riverine contaminant load will be derived directly from farmyards.

The range of contaminants that are most likely to originate from farmyards, where faecal matter is either stored or freshly deposited, is wide and inclusive of microbial/pathogenic organisms, major nutrients, potentially toxic substances (e.g., pesticides) and metabolic substrates (e.g., labile organic carbon) that contribute to biological oxygen demand. Individual farms may vary widely with respect to the range and concentrations of contaminants they generate as a consequence of the nature of the enterprise and contaminant transport pathways, such as proximity to the nearest watercourse. The presence of faecal material, whether stored or freshly deposited, represents a ready source of microbiological, nutrients and labile organic matter.

The transport and delivery of contaminants to the drainage network can be the result of episodic, hydrologically active storm events or some semi-continuous seepage perhaps from housed livestock or domestic septic overflows. An observational study by Aitken et al. (2001) of farms within a livestock (cattle) dominated area, suggested that domestic foul drainage and rainwater drainage systems were often poorly maintained, incorrectly installed and inappropriately used. The potential significance that contaminated effluent from farmyard may have upon local stream composition together with detrimental effects on aquatic macroinvertebrates has been demonstrated (Hooda et al., 2000).

The aim of this research was to sample various sources of flow generated from areas of livestock farms during summer storm events and assess their potential for delivering contaminated runoff to adjacent surface waters. This involved assessing the microbiological and chemical composition of roof and hardstanding runoff from four livestock farms sampled over a one month summer period.

2. Materials and methods

Four livestock (dairy and/or beef) enterprises situated in the River Irvine catchment area, Ayrshire SW Scotland were instrumented and intensively sampled over a one month summer period. Three farms had livestock using the hardstanding area throughout the four week sampling period from June 17th to July 13th 2002. The principle focus of this investigation was directed towards microbiological aspects of water quality due to the recent bathing beach non-compliance. The various potential sources of contamination within this general geographic area have been extensively studied (Wyer et al., 1997, 1998). The opportunity was taken to perform a chemical analysis of a sub-set of these samples.

2.1. Physical location and transport attributes

The four farms differed in their physical layout and proximity to an adjacent watercourse. Three farms (Farms 1, 2 and 4) had an obvious drainage ditch linking farmyard hardstanding drainage with a receiving tributary stream. The final farm (3) hardstanding was located in such a way that runoff formed a discrete overland flowpath which entered a first order ephemeral stream. In this case, an upstream and downstream comparison of the hardstanding was not possible and the contaminant flux at the stream monitoring point represented the loading derived from the hardstanding area.

2.2. Instrumentation

A data logging tipping bucket rain gauge (US sourced Onset$^{\mathrm{TM}}$ tipping bucket gauges recorded each 0.01 inch of rainfall) was positioned adjacent to the hardstanding of each farm. Stream and/or drain discharge measurements were completed at each farm to obtain contaminant flux data. The measurement locations were tailored to the hydrology and morphometry of each site, three sites allowed discharge measurement locations in streams below the site, the fourth farm discharge via a field drain. For the larger stream monitoring site at Farm 2, which had a clearly defined catchment of 4.9 km^2, the calculated stage discharge relationship was further checked using a rainfall runoff model (Littlewood and Jakeman, 1993).

2.3. Sample collection and analysis

The nature of the study necessitated opportunistic sampling of flows from the roofs, hardstandings and streams following rainfall events to supplement dry weather data acquisition. Duplicate samples were collected in sterile plastic containers (sterile pipettes were used to acquire shallow hardstanding drainage), and transported to the laboratory in a dark cold box. One set of samples were used for microbiological analysis while the second duplicate set of samples were stored frozen prior to their chemical analysis at the Macaulay Institute, Aberdeen. On each sampling occasion a series of five samples were collected over a 5 min period, analysed, and used to provide a single averaged value. All microbiological analyses were made on site using a mobile field laboratory where samples were immediately transferred to a dark refrigerator <4 °C and containers were in contact with melting ice to effect rapid temperature reduction. To avoid 'greater than' and 'less than' results, three serial dilutions were initially employed for all sites. Where appropriate, this was adjusted to two dilutions as the laboratory staffs were better able to predict the concentrations expected from each site. Total coliform (TC), presumptive faecal coliform (FC), and presumptive faecal streptococci (FS) were enumerated by membrane filtration using standard UK methods (Environment Agency, 2002).

After thawing of the second set of samples, total nitrogen Total-N) and total phosphorus (Total-P) were determined olorimetrically after UV oxidation (Williams et al., 1995). A sub-sample was also filtered (pore size 0.45 μm) and nalysed for total dissolved nitrogen (TDN), ammonium NH_4-N), nitrate (NO_3-N), dissolved organic nitrogen DON) was calculated by difference DON = TDN (NH_4-N + NO_3-N), total dissolved phosphorus (TDP), molybdate reactive phosphorus (MRP) dissolved organic phosphorus (DOP) was calculated by difference DOP = TDP−MRP, dissolved organic carbon (DOC) was determined using a Perkin–Elmer CHN analyzer and a range of other elements by inductively coupled plasma tomic emission spectrometry that included potassium (K), inc (Zn), silica (Si), calcium (Ca), sodium (Na), sulphur S), copper (Cu) and iron (Fe).

2.4. *Physical and management attributes of each farm*

There were significant operational and physical differences between the farms (Table 1) which provided a range of opportunities for sample collection. Only at Farm 4 were cattle not actually present (they had been removed just prior to sampling) in the farmyard during the sampling period, this location was therefore considered to provide ome indication of residual effects of stored/aged faeces/urine. Farms differed with respect to the extent and type of connectivity that existed between the farmyard and the nearest open drainage channel.

For Farm 1 this consisted of a small open ditch running along the farm access road (used daily for collecting dairy cattle) culverted through the farm which receives farmyard drain discharge from dairy washings and storm runoff. The connectivity was tested using a microbial tracer (*Serratia marcescens* bacteriophage) which demonstrated a direct connection between the sump receiving drainage from the farm hardstanding and the stream monitoring point. Tracer was evident in the stream 15 min after release into he sump. The travel distance was approximately 80 m, which indicates rapid flow through a constructed piping ystem. This ditch also probably receives seepage from the domestic septic system.

For Farm 2, runoff from the farmyard was evident during storms and this ran downhill from the farmyard to he adjacent stream along a hardcore access track. Additional runoff routes could have contributed, but none were observed even during high flow events. Farm 3 provided a good example of where runoff from the farmyard represents the main contribution to a first order stream. Runoff passes through a heavily faecal contaminated gate/collection area prior to flowing over grassland into a small ephemeral channel. At Farm 4, periodic discharge via a buried plastic field drainage pipe originating from the vacated farmyard emptied directly into an open field ditch. At each farm, any yard drainage was supplemented by roof runoff which initially discharged onto the hardstanding or joined with hardstanding drainage via a sump and pipework system.

Table 1
Broad comparison between farms

	Farm 1	Farm 2	Farm 3	Farm 4
Livestock present in farmyard	Yes	Yes	Yes	No
Direct drain connectivity to stream	Yes	Possible	Possible	Yes
Other direct connectivity to stream	Possible	Yes	Yes	Possible

2.5. *Data presentation and statistical analysis*

Five separate samples were collected from each location over a 5 min period, analysed separately and then an average of these five concentrations was used in subsequent statistical analysis. Data were analysed as a separate group on the basis of farm or time of sample collection during individual storm events. Where necessary, correlations were performed after log transformation.

3. Results

The total amount of rainfall during the study period varied between 75 and 126 mm (average 107 mm) for the individual farms which, when compared to data for the nearest meteorological station (Auchincruive) over the same period in previous years, indicates that this four week period in 2002 was certainly wetter than average (58 mm for the period 1990–1999). Samples of hardstanding runoff were broadly classified as being collected under one of five rainfall conditions (dry, drizzle, rising, peak and falling).

3.1. *Composition of roof runoff*

The averaged composition of all roof runoff samples and separated on the basis of farm are shown in Table 2. Concentrations were variable (large SE) but a significant number of samples were highly contaminated. The breakdown by individual farm suggests some definite 'local' influences acting to modify roof runoff composition. Concentrations of FS were greater than FC in all samples, being particularly noticeable at Farm 2. Runoff from Farm 4 had particularly high TDP, DOC and K concentrations, while Zn was an order of magnitude greater at both Farms 2 and 4. The relative contribution of individual N species made to TDN was essentially similar while the greatest majority (~70%) of TDP was present as DOP.

3.2. *Composition of hardstanding runoff*

Hardstanding runoff was extremely variable in its composition, concentrations of FC, FS, total N and P ranged over at least 4 orders of magnitude and the

Table 2
Mean concentration of roof runoff samples (FC and FS cfu 100 ml^{-1} and all chemical solutes mg l^{-1}) with standard error (pooled across all farms) shown in brackets

	FC	FS	TDN	TDP	DOC	K	Zn
All samples	1974 (1209)	12683 (4177)	2.52 (1.23)	0.295 (0.243)	16.6 (9.09)	18.6 (16.1)	0.218 (0.065)
Farm 1	1683	8673	0.503	0.061	7.47	3.538	0.028
Farm 2	712	26826	1.487	0.028	6.84	1.55	0.346
Farm 3	5083	12393	2.51	0.018	10.7	6.35	0.0065
Farm 4	274	837	4.57	0.913	36.6	54.5	0.252

Total number of samples was 10. Differences between individual farms are shown.

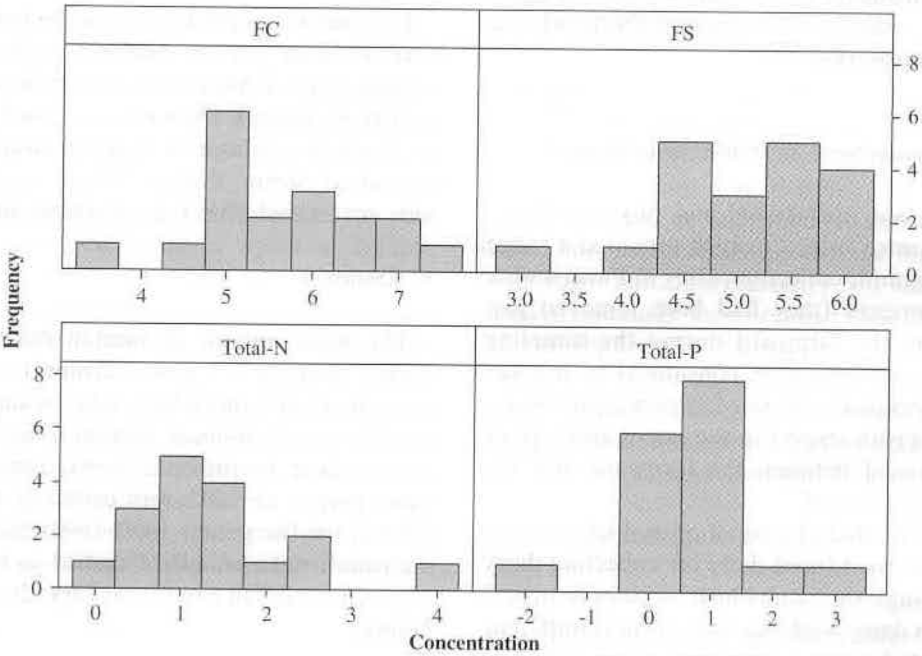


Fig. 1. Histograms showing the range of FC, FS, total N and P concentrations (cfu 100 ml^{-1} and mg N or P l^{-1}) in samples collected from hardstanding (all expressed over a log scale).

population was highly skewed (Fig. 1). A number of samples showed especially high levels of contamination (highest >8000 and 2000 mg l^{-1} of total N and P, respectively). Despite the wide range of concentrations, a highly significant relationship existed between individual components, total N and P ($p<0.001$), TC and FC ($p<0.001$) and a slightly less between FC and FS ($p<0.05$). The averaged total-N:total-P ratio was 12 and ranged between 2 and 60. Organic forms of both N and P (~60%) dominated the majority of samples.

Soluble forms of N and P represented approximately 80% of the total N and P present in unfiltered samples and demonstrated a wide variation in composition. Maximum TDN and TDP concentrations were 637 and 19.5 mg l^{-1} and a strong positive ($p<0.001$) relationship existed between each parameter (Fig. 2). The composition of TDN varied between samples but reduced forms of N (NH_4 and DON) dominated (Table 3). While NO_3 was

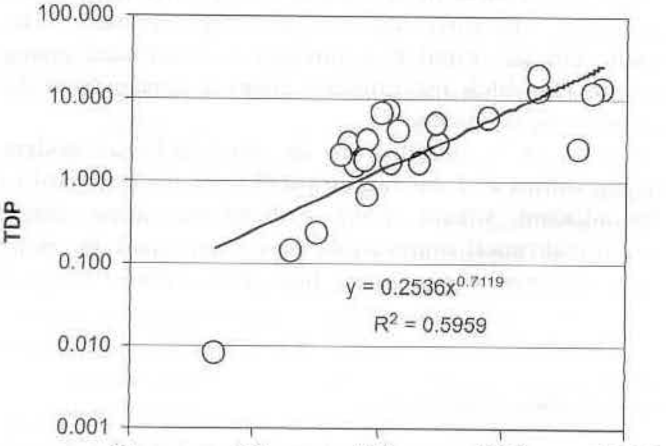


Fig. 2. Relationship between concentrations (mg l^{-1}) of TDN and TDP in samples collected from hardstandings.

present in most samples, it generally represented <15% of he TDN. An inverse relationship ($p<0.01$) between NO_3 and DON suggests that these originate from different ources. The dominant form of TDP was DRP. Concentrations of DOC and K could also be significant, and both were strongly ($p<0.001$) and positively correlated with all N and P species, except NO_3. There were no significant orrelations between any chemical forms of N and P with measured microbiological parameters. Hardstanding runoff also contained a variable cocktail of other elements, ometimes at high concentrations but showed a wide variation in composition (Fig. 3). Concentrations of Zn were variable and noticeably high in certain samples with ight samples out of 22 being greater than 0.1 mg l^{-1} with a maximum of over 0.4 mg l^{-1}.

Table 3
Concentrations (mg l^{-1}) of total and soluble major nutrient and dissolved rganic carbon of hardstanding runoff from all farms

Variable	No.	Mean	SE Mean	Minimum	Median	Maximum
TDN	22	101	39.4	0.481	12.3	637
NH_4–N	22	70.4 (48)	29.3	0.025	5.72	562
NO_3–N	22	1.24 (12)	0.665	0.008	0.063	14.28
DON	20	36.9 (45)	15.5	0.431	4.34	248
TDP	22	4.73	1.07	0.008	2.81	19.50
DRP	21	3.9 (68)	1.09	0.005	1.81	19.46
DOP	20	1.40 (39)	0.542	0.003	0.524	10.94
DOC	22	352	116	10.3	121	1894
K	22	360	109	0.098	149	1926

The average proportion of each N and P forms is shown in brackets.

3.3. Variation in composition of hardstanding runoff between farms and flow conditions

Part of the variability within the data set can be explained once data were grouped on the basis of individual farms (Table 4). The relative contributions of individual contaminants varied between the farms. For example, while runoff generated from Farm 1 was characterised by higher FC, TDN, TDP, DOC and K, concentrations of FS were an order of magnitude greater from Farm 2. Zinc was especially high in runoff from Farm 4.

Some of the remaining variability within the data set was also explained once results were expressed on the basis of the timing of sample collection during individual storm events (Fig. 4). Here the general trend appeared to differ between the determinants. A rapid decline in TDN (NH_4 and DON) occurred after the onset of a storm event, followed by a recovery as flow declined towards the end of a storm. Although reliable measurements of flow from these hardstanding is difficult and were not made, a broad

Table 4
Median values for various parameters (FC and FS cfu 100 ml^{-1}, and solutes as mg l^{-1}) in runoff from hardstanding for individual farms

Site	FC	FS	TDN	TDP	DOC	K	Zn
Farm 1	13 948 933	96 642	472	6.76	671	781	0.041
Farm 2	1 100 079	613 060	10.8	2.78	128	121	0.072
Farm 3	122 129	45 082	26.0	2.21	82.4	114	0.046
Farm 4	77 362	35 928	8.15	2.98	119	199	0.127

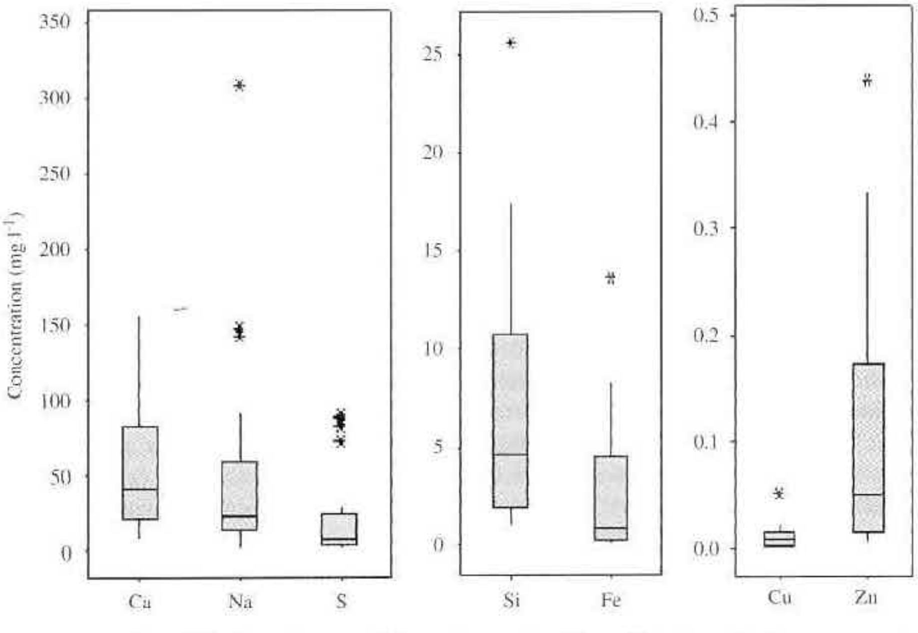


Fig. 3. Boxplots of elemental composition of hardstanding runoff (sample number, No. = 22) where the box represents roughly the middle 50% interquartile range) of the data, and lines (whiskers) indicate the general extent of the data. The horizontal line inside the box represents the median and he outliers are marked as an asterisk.

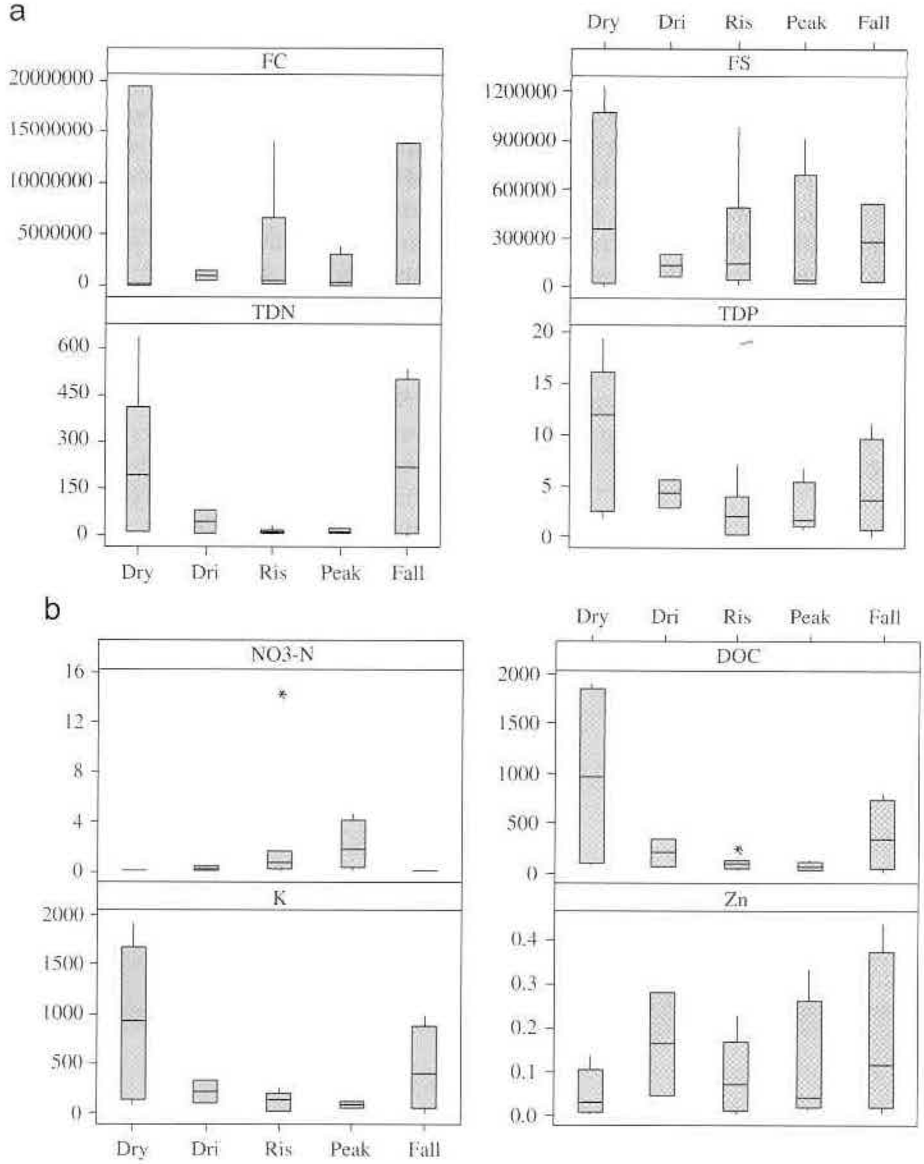


Fig. 4. Relationship between hydrological stage and concentrations (cfu 100 ml^{-1} or mg l^{-1}) of various substances in hardstanding runoff (a) FC, FS, TDN and TDP and (b) NO_3 N, DOC, K and Zn. Data are presented as boxplots where the box represents roughly the middle 50% (interquartile range) of the data, and lines (whiskers) indicate the general extent of the data. The horizontal line inside the box represents the median and the outliers are marked as an asterisk.

inverse relationship between flow and concentration for TDN (NH_4 and DON), TDP, DOC and K could be inferred using the timing of when samples were collected during the storm events. Nitrate, although present at low concentrations, tended to show an opposite trend with higher concentrations occurring in samples collected during peak storm conditions, while microbiological parameters (e.g., FC, FS Fig. 4a) showed no clear pattern.

3.4. *Relative significance of roofs and hardstanding runoff as sources of contamination*

Table 5 summarises the extent to which the composition of roof runoff is modified as it comes in contact with hardstanding areas. All determinants, except Zn and possibly NO_3, showed a relative increase in concentration between roof and hard-standing runoff, although the magnitude of any increase varied widely. The greatest relative change occurred for FC (>3 orders of magnitude) and total N and P (>2 orders of magnitude), TDN, TDP, FS and DOC (>1 order of magnitude). An indication of the overall impact that various forms of farm runoff can have on adjacent streams is also presented in Table 5. Here the average ratio of concentrations for samples taken simultaneously downstream and upstream of Farms 1 and 2 are compared for 'dry' and 'wet' conditions. Although not ideal, some confidence in this approach is provided by the very close agreement shown for Si (~ratio of 1) which is due to it having a primarily geochemical origin. Most of

Table 5
Ratio of change in concentrations between runoff from hardstanding compared to roofs averaged for all data and the ratio of averaged downstream to upstream concentrations for Farms 1 and 2 sampled under dry or wet conditions

Variable	Hardstanding/roof	Farm 1		Farm 2	
		Dry	Wet	Dry	Wet
Flow ($l\,s^{-1}$)	–	7.48	24.3	48.5	529
TC	2737	–	12.3	5.21	2.10
FC	1677	–	3.64	4.21	3.10
FS	23.5	–	2.20	3.22	0.78
Total–N	131	2.77	3.70	1.94	1.00
Total–P	171	4.48	5.46	3.10	0.76
TDN	40.1	6.17	1.90	1.92	1.00
NH_4–N	44.0	88.2	8.82	33.7	1.13
NO_3–N	2.40	0.05	3.02	1.38	0.94
DON	75.3	20.1	1.26	1.70	1.02
TDP	16.0	21.4	1.75	5.10	1.31
PO_4–P	19.2	40.3	3.39	6.30	1.41
DOP	5.30	13.1	0.18	4.40	1.18
DOC	21.2	1.92	1.44	1.26	1.03
K	19.4	16.4	10.4	3.74	1.16
Zn	0.50	13.3	2.23	4.31	1.94
Si		1.08	1.08	1.05	1.13

Averages calculated from a minimum of three individual samples and '–' indicates insufficient data.

the potential contaminants demonstrate an increase below each farm, which is especially noticeable for NH_4, DON and TDP under dry conditions. All microbiological parameters (except FS wet flow, Farm 2) showed a ratio >1 during both dry and wet flows. Total N and P suggested increases of between 2 and 5 times under both dry and wet conditions at Farm 1, but only dry conditions at Farm 2. The extremely high 'wet' flows at Farm 2 mean that despite showing only relatively modest increased ratios (few values >1.5), these actually represent considerable increase in loads.

The geographical layout at Farm 3 being located at the head of an ephemeral, first order stream, makes it possible to estimate actual loadings arising from the farmyard area (Table 6). Here small flows under dry conditions prior to a storm event showed particularly high concentrations of TDN and K. Subsequently samples were collected under increased flow (rising and peak) conditions where concentrations of TDN and K declined while FC, FS and TDP increased. The composition of TDN and TDP present in stream flow dominated by NH_4 and DOP under dry conditions while the relative contribution made by NO_3 and PO_4 increased considerably under wet conditions.

Table 6
Change in concentrations of FC (cfu $100\,ml^{-1}$) and nutrients ($mg\,l^{-1}$) through a single storm on the 30/6/02 for roof, hardstanding and open channel flow at farm 3

	Pre-storm	Rising	Peak
Flow ($l\,s^{-1}$)	0.018	0.157	0.741
Roof runoff			
FC		9129	1037
FS		16 561	8224
TDN		4.39	0.64
TDP		0.031	0.005
K		12.29	0.40
Si		2.01	1.02
Hardstanding runoff			
FC		106 091	122 129
FS		11 556	29 600
TDN		3.24	8.28
TDP		0.229	0.675
K		10.39	48.37
Si		2.04	1.36
Stream			
FC	98 355	240 785	478 322
FS	6979	31 993	42 557
TDN	34.56 (84, 2.0)	10.09 (32, 55)	7.90 (43, 44)
TDP	0.128 (25)	0.413 (57)	0.806 (70)
K	83.89	25.55	37.55
Si	6.39	5.33	3.76

The percent NH_4–N and NO_3–N and PO_4–P for the stream samples are shown in brackets.

4. Discussion

The analysis of runoff generated from roof and hardstanding demonstrated variable but often high concentrations of multiple groups of contaminants. It is difficult to predict how results collected for this summer sampling period is directly transferable to other seasons, however, data collected by Hooda et al. (2000) suggest continuous local impacts. Livestock numbers within the farmyard are likely to vary especially where cattle are housed over winter. A more regular number of livestock may be expected on dairy farms, where twice daily movement to and from milking parlours/collecting/feeding areas provides a high risk. Other periods such as clearing out of housed cattle bedding or slurry stores representative of other operations during which hardstanding can become contaminated with faeces.

The data in Table 6 demonstrate that while roof runoff appears to show declining concentrations of contaminants as the storm progresses, hardstanding runoff remains highly contaminated. This suggests that the stores of contaminated roof material (e.g., bird droppings) are limited and become easily exhausted during individual storms and require some 'recovery' period between events. A similar situation does not seem to arise for runoff generated from hardstanding as early and late storm runoff is equally contaminated. The above average precipitation volume during the sampling period compared with the previous 10 year average, suggests that even during wet periods farmyard hardstanding continues to represent a significant potential source of a wide range of chemical and microbiological contaminants. There was also sufficient residual faecal material even at Farm 4, where livestock were not present during the sampling period, to produce contaminated runoff.

Information describing the quality of roof runoff is uncommon; here we demonstrate that this may well represent a significant source of contamination. There are various possible sources for these contaminants which include bird droppings, atmospheric dry deposition which deposit and accumulate onto roof surface. Significant concentrations of FS, TDP and K probably reflect this situation. Part of the N may well originate from gaseous sources, such as ammonia volatilised from local urine/faecal sources. Roof runoff concentrations declined with time during an individual storm event (Table 6) which might be considered as a 'source limited' situation and washout of contaminants. One further roof source is demonstrated by concentrations of Zn which is high at two farms and presumably originates from galvanised surfaces. Storm frequency as well as availability of grain as bird food are attributes that might be expected to influence runoff composition. Conditions that enhance corrosion of galvanised surfaces might have resulted from the acidity generated locally through emissions of NH_3 (derived from livestock wastes) and the eventual dry or wet deposition as $(NH_4)_2SO_4$ where SO_4 produces H_2SO_4, and $2NH_4$ oxidation (nitrification) produces 4H. Other studies that calculated elemental balances have also demonstrated that farmyards appear to represent a significant source of Zn (Bengtsson et al., 2003).

The route of roof runoff varied between farms, while some guttering/down pipes were missing or broken other down pipes emptied directly onto hardstanding. At each location, direct hardstanding runoff was supplemented, to varying extents, by contaminated roof flow. Possible sources of contaminants, include fresh and stored livestock faeces/urine (slurry and FYM), dairy/yard/machinery washings and domestic waste if connected to household septic systems. Roof runoff generally had smaller FC concentrations than harstanding flows, which had been in contact with livestock faeces but it did exhibit higher FS concentrations than some of the streams receiving this combination of input sources. Areas of hardstanding that usually form integral components of many farmyards possess certain attributes that are likely to make them potential sources of contaminants. A comparison of the average ratio of hardstanding to roof runoff concentrations demonstrates the relative significance of individual sources for the various contaminants. Those contaminants most attributable to faeces and urine as a source include TC, FC, total and dissolved N and P, DOC and K demonstrate large shifts in concentration. In contrast FS, and NO_3 show a smaller relative increase and Zn concentrations are even diluted.

The chemical forms of N and P is an important property when assessing possible environmental significance. A large proportion of the N and P in roof runoff was present as dissolved forms (65% and 20%, respectively). The predominant fraction (70%) of the TDN in roof runoff was present as inorganic forms, while <25% of TDP was DRP. The proportions of NO_3-N and NH_4-N were similar in roof runoff, however, hardstanding runoff was dominated (60%) by DON and DOP in TN and P. DRP dominates TDP, NH_4-N and DON dominated TDN (~90%). An indication of the bioavailability or lability of the organic forms is the DOC:DON ratio, which was 50 for roof runoff compared to 26 for hardstanding runoff.

The data clearly demonstrate that farmyards represent a long-term source for a wide range of contaminants and measurable biological impacts have been detected (Hooda et al., 2000). The greatest opportunity for actual transfer of contaminants would be during high flows where there is the possibility for source limitation. Seepage (for example from slurry stores or sheds) during dry periods or light drizzle has the capacity to move contaminants which may collect somewhere along the transport route. The most obvious periods of transport and delivery of contaminants to adjacent drainage ditches occurs during individual storm events. While there is some evidence from the data that a reduction in concentrations occur under peak flows this may reflect some source limitation or simply reflect dilution of grossly contaminated flows by relatively less contaminated runoff from 'clean' areas. A second, less obvious but equally important phase of transport may occur during either dry or light drizzle periods. Despite the smaller flows any runoff that occurs under these conditions may result in production of effluent with relatively high concentrations, possibly even direct seepage from housing areas. The low permeability of these areas means they can become hydrologically active during very light storm events when other more permeable surface show a limited response (Hively et al., 2005). Of more significance might be the partial transport off-site resulting in the supplementation of 'receiving' areas, including gateways. These areas become source areas during hydrologically active periods, supplementing early runoff fluxes. It is also possible, depending upon the contaminant involved, that these collecting areas see various transformations, which may involve, microbiological die-off and transformation of nutrient chemical forms (e.g., nitrification or denitrification). The behaviour of individual contaminants within the environment differs which introduces a degree of selectivity in terms of transport and environmental impact.

Farmyards can contribute significantly to catchment nutrient and faecal indicator organisms (FIO) outputs when they have

(1) a ready and renewable source of faecal material and/or agrochemicals is present,
(2) a direct hydrological connection with open water channels exists, and
(3) a sufficient proportion of livestock farms are present in the catchment.

5. Conclusions

Runoff generated by summer storms from areas of farm hardstanding exposed to livestock is variable in composition but could have potentially high concentrations of microbiological agents, major nutrients and metals (e.g., Zn). Where hardstanding runoff reaches and makes a significant contribution to surface waters it has the potential to impact water quality, especially in local headwater streams. The composition of runoff varied between individual farms and also with time during individual storm events. Reduced forms of N (NH_4 and DON) and DRP dominated runoff samples and these together with DOC and FIO were all well correlated with each other indicating a common source. Separate analysis of roof runoff demonstrated this could represent a significant source of FS and Zn, and where this runoff complements and mixes with direct hardstanding runoff it becomes exposed to further contamination by livestock faeces and urine. The simple division of samples based upon time of collection during individual storms suggests a source controlled limitation coupled with some form of 'dilution' factor caused by roof runoff. Frequent renewal of faeces and urine materials where livestock are present mean that sources can be replaced out with storms. The repeated production of contaminated hardstanding runoff during consecutive storm events suggested that farmyards represent a substantial long-term source of potential contaminants. All farmyards showed a potential for a dynamic and rapid linkage with adjacent surface waters, even during relatively light rainfall events.

Acknowledgements

This work was funded by the Scottish Executive Environmental and Rural Affairs Department. The help of National Farmers Union, Scotland, local SEPA staff and four farmers who allowed access and instrumentation of their farmyards is acknowledged. Yvonne Cook, Macaulay Institute, organised and carried out all the nutrient analysis.

References

Aitken, M., Merrilees, D., Duncan, A., 2001. Impact of Agricultural Practices and Catchment Characteristics on Ayrshire Bathing Waters. Scottish Executive Central Research Unit. Available from: ⟨www.scotland.gov.uk/Resource/Doc/1057/0012202.pdf⟩.

Bengtsson, H., Öborn, I., Jonsson, S., Nilsson, I., Andersson, A., 2003. Field balances of some mineral nutrients and trace elements in organic and conventional dairy farming – a case study at Öjebyn, Sweden. European Journal of Agronomy 20, 101–116.

Environment Agency, 2002. The Microbiology of Drinking Water (2002). Methods for the Examination of Waters and Associated Materials. Standing Committee of Analysts.

Dunne, E.J., Culleton, N., O'Donovan, G., Harrington, R., Olsen, A.E., 2005. An integrated constructed wetland to treat contaminants and nutrients from dairy farmyard dirty water. Ecological Engineering 24, 221–234.

Hively, W.D., Bryant, R.B., Fahey, T.J., 2005. Phosphorus concentrations in overland flow from diverse locations on a New York dairy farm. Journal of Environmental Quality 34, 1224–1233.

Hooda, P.S., Moynagh, M., Svoboda, I., Miller, A., 2000. Macroinvertebrate as bioindicators of water pollution in streams draining dairy farm catchments. Chemistry and Ecology 17, 17–30.

Littlewood, I.G., Jakeman, A.J., 1993. Characterisation of quick and slow stream flow components by unit hydrograph for single and multi-basin studies. In: Robinson, M. (Ed.), Methods of Hydrological Basin Comparison. IOH Report 120, pp. 99–111.

Misselbrook, T.H., Webb, J., Chadwick, D.R., Ellis, S., Pain, B.F., 2001. Agricultural hardstandings used by livestock – a previously unquantified source of ammonia emissions. Atmospheric Environment 35, 5331–5338.

Neumann, M., Schulz, R., Schafer, K., Muller, W., Mannheller, W., Liess, M., 2002. The significance of entry routes as point and non-point sources of pesticides in small streams. Water Research 36, 835–842.

Williams, B.L., Shand, C.A., Hill, M., O'Hara, C., Smith, S., Young, M.E., 1995. A procedure for the simultaneous oxidation of total soluble nitrogen and phosphorus in extracts of fresh and fumigated soils and litters. Communications in Soil Science and Plant Analysis 26, 91–106.

Wyer, M.D., Kay, D., Crowther, J., O'Neill, J.G., Jackson, G., Fewtrell, L., 1997. The effects of catchment (i.e. non-outfall) sources of faecal indicators on compliance of marine recreational waters with directive 76/160/EEC. Water Science and Technology 35 (11–12), 151–156.

Wyer, M., Kay, D., Crowther, J., Whittle, J., Spence, A., Huen, V., Wilson, C., Carbo, P., Newsome, J., 1998. Faecal indicator budgets for recreational coastal waters – a catchment approach. Journal of the Chartered Institution of Water and Environmental Management 12 (6), 414–424.

ELSEVIER

Journal of Environmental Management 87 (2008) 560–566

Journal of
Environmental
Management

www.elsevier.com/locate/jenvman

Stormwater runoff and export changes with development in a traditional and low impact subdivision

Michael E. Dietz[a,*], John C. Clausen[b]

[a]*Department of Environment and Society, 5215 Old Main Hill, Utah State University, Logan, UT 84322 5215, USA*
[b]*Department of Natural Resources Management and Engineering, University of Connecticut, 1376 Storrs Rd., Unit 4087 Storrs, CT 06269-4087, USA*

Received 26 September 2006; received in revised form 30 January 2007; accepted 10 March 2007

Abstract

Development continues at a rapid pace throughout the country. Runoff from the impervious surfaces in these watersheds continues to be a major cause of degradation to freshwater bodies and estuaries. Low impact development techniques have been recommended to reduce these impacts. In this study, stormwater runoff and pollutant concentrations were measured as development progressed in both a traditional development, and a development that used low impact development techniques. Increases in total impervious area in each watershed were also measured. Regression relationships were developed between total impervious area and stormwater runoff/pollutant export. Significant, logarithmic increases in stormwater runoff and nitrogen and phosphorus export were found as development occurred in the traditional subdivision. The increases in stormwater runoff and pollutant export were more than two orders of magnitude. TN and TP export after development was 10 and 1 $\mathrm{kg\,ha^{-1}\,yr^{-1}}$, respectively, which was consistent with export from other urban/developed areas. In contrast, stormwater runoff and pollutant export from the low impact subdivision remained unchanged from pre-development levels. TN and TP export from the low impact subdivision were consistent with export values from forested watersheds. The results of this study indicate that the use of low impact development techniques on a watershed scale can greatly reduce the impacts of development on local waterways.

Keywords: Stormwater runoff; Impervious; Export; Low impact development; Nonpoint pollution

---

1. Introduction

Runoff from developed areas continues to be a leading cause of impairments in the nation's waterways (US EPA, 2002). Development continues at a rapid pace throughout the country, with some cities increasing in size by up to 50% in the past 30 years (US EPA, 2001). Several research studies have documented increases in runoff volume (Jennings and Jarnagin, 2002; Waananen, 1969) and peak flow rates (Leopold, 1968) as areas were transformed from undeveloped to urban. Other studies involving computer modeling of future increases in impervious areas have also predicted increased runoff volumes (Hollis, 1977; James, 1965; Pawlow and Nathan, 1977; Sloto, 1988). In addition, numerous studies have documented decreased water quality in urban runoff (Makepeace et al., 1995).

Imperviousness has been recommended as an indicator for stream health (Arnold and Gibbons, 1996). A variety of impacts have been associated with increased impervious cover, including decreased fish species richness and abundance (Wang et al., 2001), channel morphology changes (Booth et al., 2002), decreased benthic organism richness (Roy et al., 2003) and abundance (Klein, 1979), decreased base flow in streams (Ferguson and Suckling, 1990; Wang et al., 2001), and decreased water quality (Carle et al., 2005; Roy et al., 2003). More complex predictors of stream impacts such as the multimetric urban index composed of numerous infrastructure, socioeconomic, and land cover variables have been proposed (Coles et al., 2004). However, total percent impervious area was found to correlate highly ($R^2 = 0.96$) with the urban index (Coles et al., 2004). This suggests that percent

---

[*] Corresponding author. Tel.: +1 435 797 3313; fax: +1 435 797 4048.
E-mail addresses: michael.dietz@usu.edu (M.E. Dietz), john.clausen@uconn.edu (J.C. Clausen).

0301-4797/$ - see front matter
doi:10.1016/j.jenvman.2007.03.026

impervious area is valid as a predictor of stream impacts, and it is a simpler indicator to use.

A degradation threshold value at about 10% imperviousness has been cited by several authors (Booth and Reinelt, 1993; Klein, 1979; Schueler, 1994, 2003; Wang et al., 2001). Watersheds with low levels of imperviousness may have a broad range of responses due to complex watershed interactions, but highly developed watersheds have uniformly poor conditions (Booth et al., 2004; Wang et al., 2001). Interpretation of threshold values in the literature should be done carefully due to the use of different measurement methods (Brabec et al., 2002). However, a definite relationship appears to exist between impervious area and multiple measures of stream health.

Recent advances in stormwater management, including low impact development (LID) techniques (Prince George's County, 1999), have provided engineers with a variety of tools to use in place of traditional catch basins and detention ponds. The overall goal of LID is to mimic the predevelopment hydrology of an area, including the runoff volumes that existed before development. Current stormwater design in most municipalities mitigates peak flow rates, but does not address the increases in stormwater volume associated with development. Cluster designs, grassed swales, rain gardens, and pervious pavements all contribute to a reduced overall impervious footprint, and encourage decentralized treatment and infiltration of stormwater runoff. Research on individual LID practices shows that pollutant attenuation, reduced flow volumes, and reduced peak flow rates can occur (Davis et al., 2001; Dietz and Clausen, 2005, 2006; US EPA, 2000). However, there is a lack of peer-reviewed studies demonstrating the effectiveness of the use of LID on a watershed scale.

Although some studies have documented increases in runoff volume as an area was developed, much of the recent research relates to the comparison of different watersheds with varying land uses. Although the information provided by such studies is valuable, it is more difficult to establish causality when data from different watersheds are analyzed at a discreet point in time. Other confounding factors such as different monitoring methods, watershed characteristics, and weather variations can make comparisons difficult. Computer modeling studies can also provide insight into potential impacts to water resources, but simplifying assumptions are often made to calibrate models, which can make it difficult to determine the significance of the results. The objective of this study was to compare stormwater runoff volume and pollutant export from adjacent traditional and LID subdivisions, as development occurred, and as impervious surfaces were added in each of the watersheds.

2. Methods

2.1. Study area

The project was located in the town of Waterford, CT, in a drainage basin contributing to a small estuary called Jordan Cove, which discharges into the Long Island Sound. The "traditional" site was a 2.0 ha subdivision containing 17 lots (Fig. 1), which was built using current regulations and construction practices. Traditional zoning was used, as was a curb and gutter stormwater collection system. A typical 8.5-m asphalt road was installed. Landscaping and turf are similar to other new subdivisions. Roof runoff was directed to lawn areas or onto driveways. Erosion and sediment controls used during construction were typical of other construction sites statewide. Construction in the traditional subdivision began in 1997, and continued through 2003. Total impervious surface coverage after construction was 32%.

The 1.7 ha LID subdivision had 12 lots (Fig. 2). Several pollution prevention measures were incorporated as part of its design. A main feature was the replacement of a traditional 8.5 m asphalt road and associated curb and gutter stormwater collection system, with a 6.1 m wide Ecostone® paver road and grassed swales. A bioretention cul-de-sac that allowed for detention and infiltration of runoff was constructed in lieu of a conventional paved area. Individual bioretention areas (rain gardens) were incorporated into each lot to detain and infiltrate roof and lot runoff. Two shared driveways and one individual driveway used traditional asphalt paving. Four driveways were constructed using alternatives to traditional asphalt: two shared driveways used Ecostone® pavers; one shared driveway and one single driveway used crushed stone (Gilbert and Clausen, 2006). Houses were constructed in a cluster layout with reduced lawns and low-mow areas. Deed restrictions were developed to prevent certain activities during the study, such as filling in of rain gardens or swales, and the addition of more impervious surface to a lot. Ongoing education programs were used to instruct owners on good housekeeping practices. Additional best management practices (BMPs) were used during construction, including locating and seeding stockpiles to prevent sediment loss, hay bales, silt fence, earthen berms, and post-storm maintenance. Construction in the LID subdivision began in 1999, and continued through 2002. After completion, total impervious area was 21%.

The project was located in a climate that is influenced by both continental polar and maritime tropical air masses (Brumbach, 1965). Average annual precipitation is approximately 1237 mm and is distributed uniformly throughout the year. Hurricanes enter the state periodically. Soils on the sites were mapped as Canton and Charlton (mesic typic Dystrudepts). The typical infiltration rate for this type of soil is 33 cm hr^{-1} (USDA, 2007).

2.2. Monitoring

Stormwater volume in the traditional subdivision was measured using an ISCO 4230 bubbler flow meter and a 38.1 cm Palmer–Bowlus flume attached to a stormwater

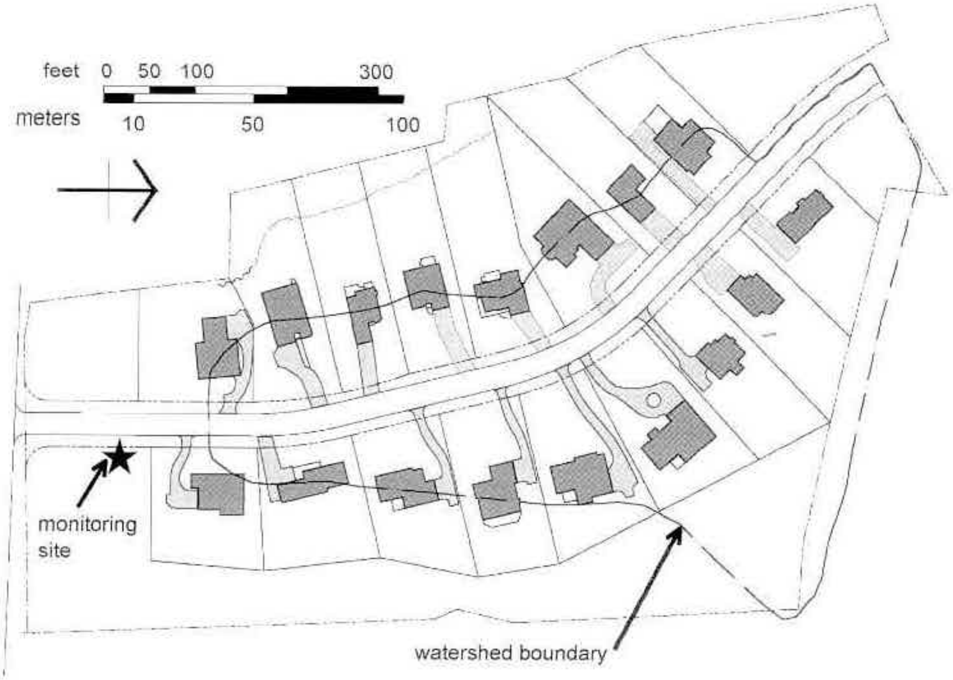


Fig. 1. Traditional subdivision layout in Waterford, CT.

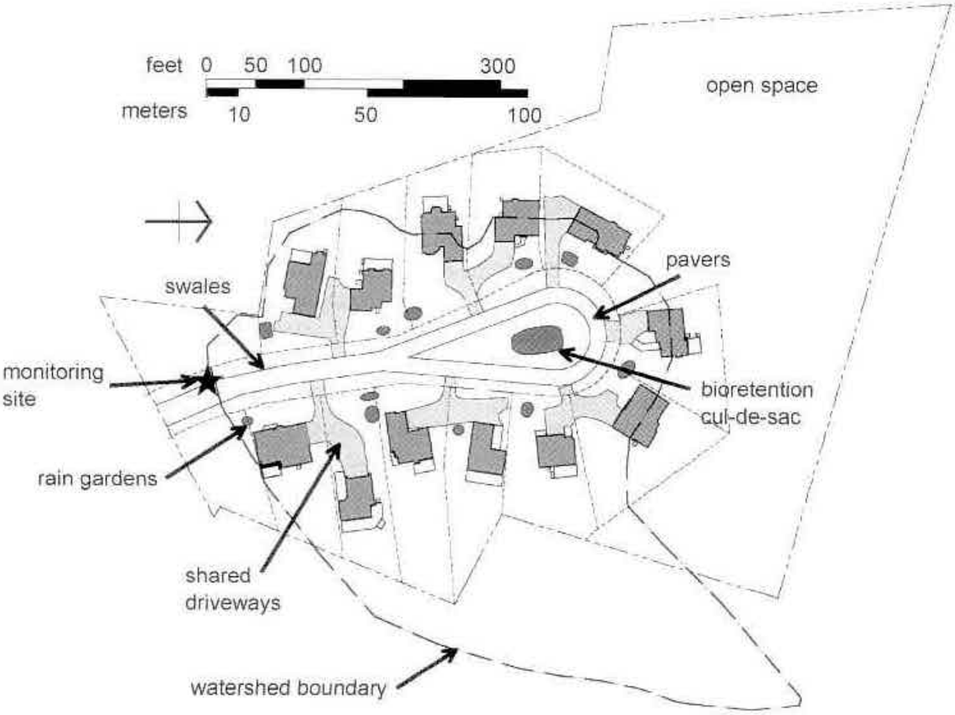


Fig. 2. LID subdivision layout in Waterford, CT.

pipe. Stormwater volume in the LID subdivision was measured using an ISCO 4230 flow meter and a 45.7 cm H-flume located at the end of a grassed swale.

Flow-weighted samples were collected automatically by an ISCO sampler, and were refrigerated in situ. Weekly samples were immediately placed in a cooler with ice packs and transported to the water quality laboratory where they were stored in a refrigerator at a constant temperature of 4 °C.

Due to an inconsistent precipitation record at the study site, monthly precipitation data from the National Climatic Data Center in Groton, CT (station #063207), which is approximately 6 km from the study site, was used as a reference (NOAA, 2006).

2.3. Sample analysis

Acidified composite stormwater samples were analyzed for nitrate + nitrite nitrogen (NO_3–N), ammonia nitrogen (NH_3–N), total kjeldahl nitrogen (TKN), and total phosphorus (TP) using a Lachat™ colorimetric flow injection system (US EPA, 1983a). Mass export ($kg\,ha^{-1}\,yr^{-1}$) was calculated by multiplying weekly cumulative flow by weekly sample concentration values, dividing by the watershed area, and summing for the year. Total nitrogen (TN) values were calculated by summing TKN and NO_3-N mass export values.

2.4. Impervious area calculation

A weekly field log was maintained on construction activities in both subdivisions, in which installation dates for driveways and roads were documented. Impervious area was calculated by hand measurements in the field. A house was considered impervious area when the roof was installed. The percent impervious of the subdivision was calculated based on total impervious area present on a weekly basis, divided by the total watershed area. An annual average of weekly percent impervious area values was calculated. Sidewalks and patios were a minute part of both watersheds, and were not included in percent impervious calculations.

Due to changes in the disturbed area on the construction site, water flow paths were altered during construction. As a result, the watershed areas for the traditional and LID sites varied during land development. Although the amount of impervious area increased continuously until completion, the overall watershed area may have been higher or lower than the previous year. Therefore, the total impervious area percentage for a given year may be higher or lower than the previous year.

2.5. Data analysis

Flow volume and pollutant export were summarized for each year, for each subdivision. Average total impervious area (%) for each year was also calculated for each subdivision. A log-normal relationship was then developed for each subdivision, with the independent variable being watershed impervious coverage (%), and yearly flow or pollutant export values being dependent variables. Each point on the graphs therefore represents a year, from 1996 through 2004. Runoff coefficients for each year were calculated by dividing annual runoff by annual precipitation, and multiplying by 100. Regression significance testing, R^2 calculations, and parameter estimates were performed in JMP (JMP, 2002) statistical package, version 5.1.

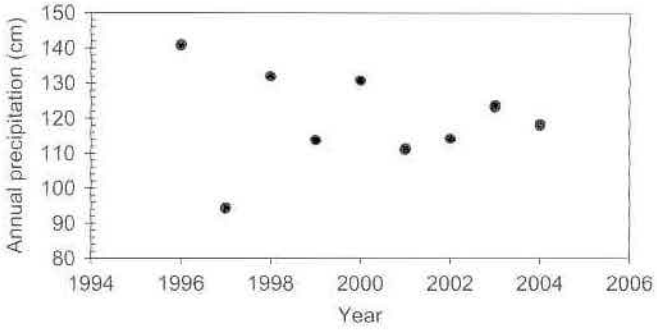


Fig. 3. Annual precipitation totals, 1996–2004, Groton, CT.

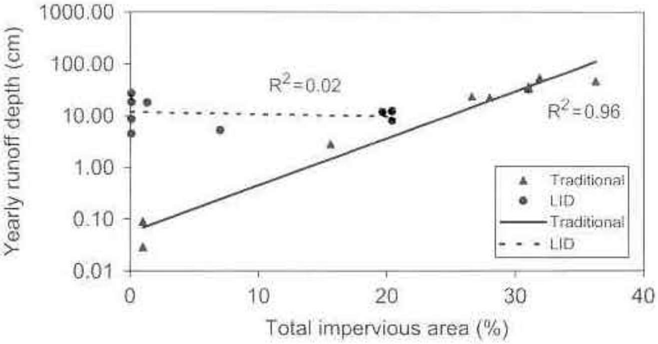


Fig. 4. Annual runoff depth vs. total impervious area, traditional and LID subdivision, 1996–2004.

3. Results and Discussion

3.1. Precipitation

Annual precipitation varied from 14% above normal in 1996 to 24% below normal 1997 (Fig. 3). For other years, variation was 10% or less of the 30-year normal precipitation (123.8 cm).

3.2. Stormwater runoff volume

Changes in stormwater volume were found as total impervious area increased in the traditional subdivision (Fig. 4). As impervious area increased from 1% to about 32%, annual runoff increased 49,000% from 0.1 cm to over 50 cm, or more than two orders of magnitude. Since precipitation during this period followed no trend, this change was due to the development of the subdivision. This regression was significant ($p = 0.001$) and logarithmic, indicating an exponential increase in stormwater volume as impervious area was added (Fig. 4). A similar exponential increase in the runoff coefficients was also found as watershed impervious area increased in the traditional subdivision (Fig. 5). The maximum runoff coefficient in the traditional subdivision was 47%, and within the range of coefficients reported by others (Novotny and Olem, 1994; Schueler, 1994).

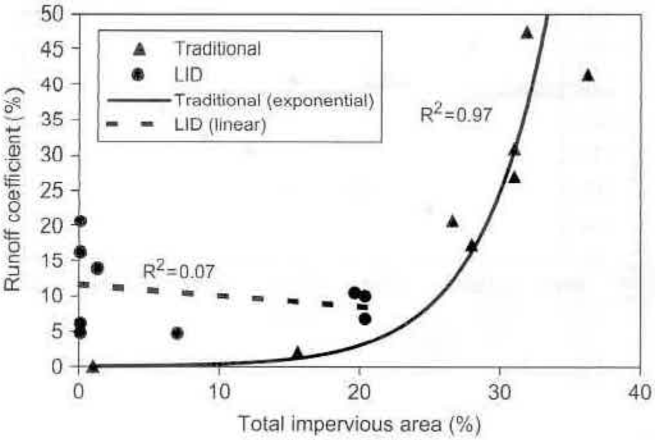


Fig. 5. Total impervious area vs. runoff coefficient, traditional and LID subdivision, 1996–2004.

Other researchers have documented stormwater volume increases of 100% (Jennings and Jarnagin, 2002) and 500% (Waananen, 1969) as impervious coverage increased in a watershed. One modeling study showed increases in runoff volume up to 12,400% as total impervious area increased 45% (James, 1965), although this increase is not typical of other values in the literature. The more dramatic increase in runoff volume found in the current study may be a result of scale: the study watershed in Waterford was 1.7 ha, whereas the catchments of the studies previously mentioned were 1320 ha (Waananen, 1969) and 6100 ha (Jennings and Jarnagin, 2002) in size. As watersheds increase in size, streamflow response (per unit area) to an event tends to become more dampened (Dunne and Leopold, 1978). Small watersheds also respond more quickly to an event, or have a shorter time of concentration. Therefore, modifications in a small watershed will result in more prominent flow changes than if similar changes were made in a large watershed. The current study shows that the impact of increased stormwater runoff on local streams due to changes in a smaller watershed can be dramatic.

Due to differences in topography and soils, the LID subdivision had more runoff before development than the traditional watershed (Fig. 4). Despite this initial difference, runoff volume and runoff coefficients in the LID watershed did not change as impervious area increased from zero to 21% (Figs. 4 and 5). A non-significant regression for the LID watershed confirms the lack of a relationship. The flow increases noted in other studies with similar increases in impervious area (Hollis, 1977; Waananen, 1969) were not found in this LID subdivision. This finding can only be attributed to the LID stormwater management techniques distributed throughout this watershed.

3.3. Nutrient export

Nutrient export showed a similar response to runoff volume. NO_3–N export increased logarithmically in the traditional subdivision with development, however no change was found in the LID subdivision (Fig. 6a). NH_3–N export from the traditional subdivision was similar to NO_3–N export, however, for the LID subdivision, NH_3-N export actually significantly decreased ($p = 0.05$) with increasing impervious area (Fig. 6b).

The change in TN export was similar to the change in NO_3–N export, with a significant logarithmic relationship for the traditional subdivision, and no relationship for the LID subdivision (Fig. 6c). TN export values for the traditional subdivision after development were approximately $10\,\mathrm{kg\,ha^{-1}\,yr^{-1}}$ (Fig. 6c). Average TN export in an urban watershed (1999–2001) with 27% impervious area in Maryland was $8.6\,\mathrm{kg\,ha^{-1}\,yr^{-1}}$ (Groffman et al., 2004). Medium density urban watersheds around the country were found to have a mean TN export of $9.6\,\mathrm{kg\,ha^{-1}\,yr^{-1}}$ (US EPA, 1983b). Increases in development in North Carolina have been found to cause significantly higher TN export (Atasoy et al., 2006). In contrast, TN export from the LID watershed averaged $2\,\mathrm{kg\,ha^{-1}\,yr^{-1}}$, which is similar to TN export from forested watersheds (Frink, 1991). TN export from three urban/suburban watersheds in Maryland (1999–2001) with impervious area similar to that of the LID watershed was much higher at $6.0–7.4\,\mathrm{kg\,ha^{-1}\,yr^{-1}}$ (Groffman et al., 2004).

TP export was similar to nitrogen: a significant ($p = 0.001$) logarithmic trend was found for the traditional subdivision, whereas no trend was found for the LID watershed (Fig. 6d). After development, TP export from the traditional subdivision was approximately $2\,\mathrm{kg\,ha^{-1}\,yr^{-1}}$ (Fig. 6d). TP export from medium density and high density urban areas was found by EPA to be 1.48 and $2.45\,\mathrm{kg\,ha^{-1}\,yr^{-1}}$, respectively (US EPA, 1983b). In contrast, average TP export from the LID subdivision was $0.4\,\mathrm{kg\,ha^{-1}\,yr^{-1}}$, which is much lower than the 0.99 and $1.48\,\mathrm{kg\,ha^{-1}\,yr^{-1}}$ reported for low density and high density urban areas in the United States, respectively (US EPA, 1983b).

4. Conclusions

A large increase in runoff volume was observed as total impervious area increased through development of a traditional subdivision in Waterford, CT. Runoff coefficients also increased. These relationships were non-linear, indicating that as imperviousness increases, annual stormwater runoff volume increases exponentially. In contrast, annual stormwater runoff volume in the LID subdivision did not change as watershed impervious coverage increased. This lack of change in flow with increased impervious area is attributed to the LID stormwater management techniques used throughout the watershed.

Pollutant export regressions were similar to runoff regressions, indicating that the flow increase in the traditional subdivision was the primary driver behind pollutant export increases. In general, pollutant export from the traditional subdivision was in line with export

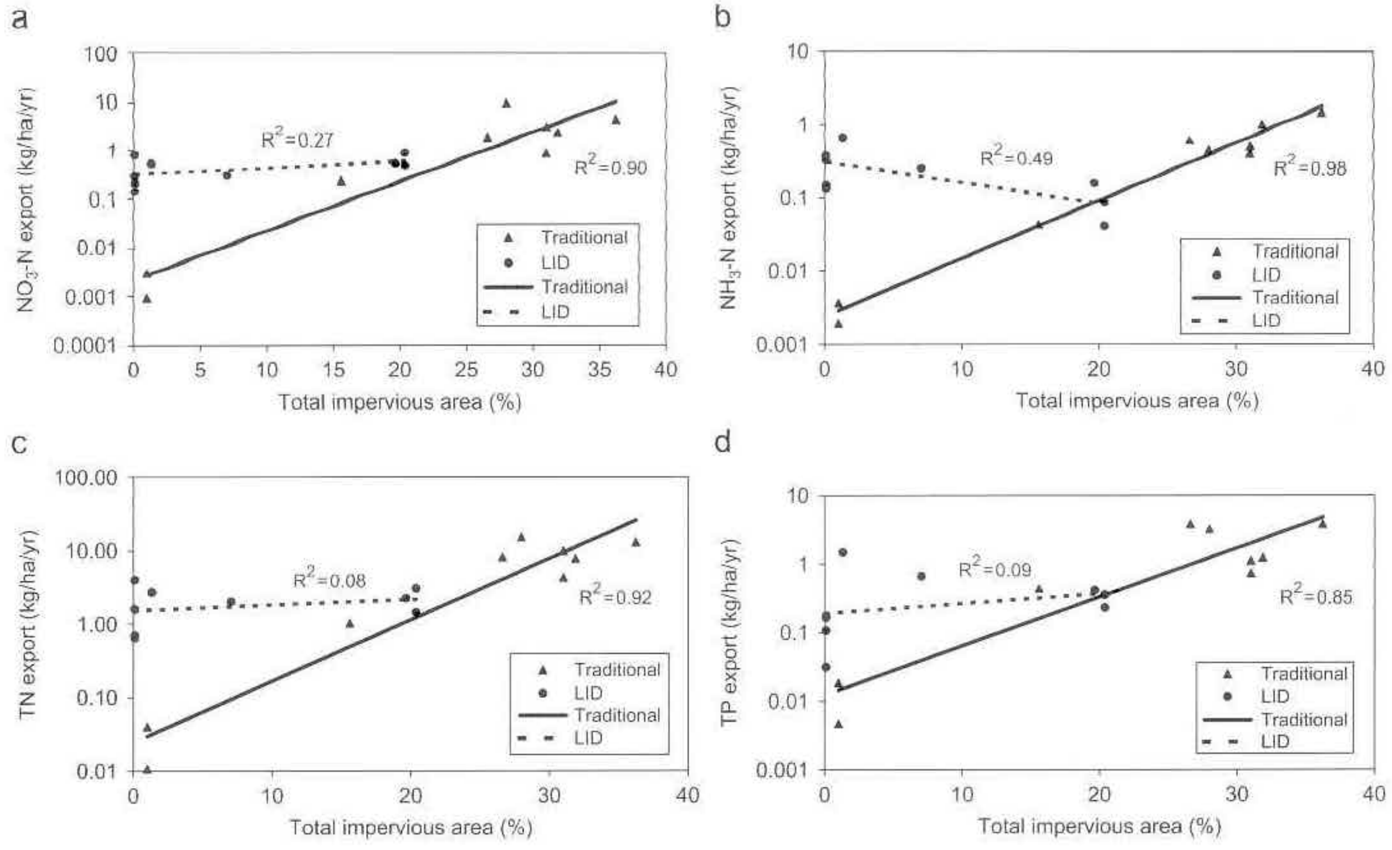


Fig. 6. Nutrient export (1996–2004) from traditional and LID subdivision: (a) NO_3–N, (b) NH_3–N, (c) TN, and (d) TP.

from urbanized watersheds, whereas pollutant export from the LID subdivision was more consistent with export from forested watersheds.

This paper did not examine peak flow rates or the responses of the different subdivisions to extreme events. The focus was the impact of the LID approach on the annual hydrologic budget. These findings indicate that the use of LID techniques on a watershed scale can significantly reduce the impacts of development on downstream water bodies.

Acknowledgements

This study was funded in part by the CT DEP through a US EPA nonpoint source grant under Section 319 of the Clean Water Act. The authors would also like to thank John Alexopoulos at the University of Connecticut for creating the subdivision drawings.

References

Arnold Jr., C.L., Gibbons, C.J., 1996. Impervious surface coverage: the emergence of a key environmental indicator. Journal of the American Planning Association 62 (2), 243–258.

Atasoy, M., Palmquist, R.B., Phaneuf, D.J., 2006. Estimating the effects of urban residential development on water quality using microdata. Journal of Environmental Management 79 (4), 399–408.

Booth, D.B., and Reinelt, L.E., 1993. Consequences of urbanization on aquatic systems-measured effects, degradation thresholds, and corrective strategies. Proceedings of Watershed '93: A National Conference on Watershed Management, March 21–24, Alexandria, VA.

Booth, D.B., Hartley, D., Jackson, R., 2002. Forest cover, impervious-surface area, and the mitigation of stormwater impacts. Journal of the American Water Resources Association 38 (3), 835–846.

Booth, D.B., Karr, J.R., Schauman, S., Konrad, C.P., Morley, S.A., Larson, M.G., Burges, S.J., 2004. Reviving urban streams: land use, hydrology, biology, and human behavior. Journal of the American Water Resources Association 40 (5), 1351–1364.

Brabec, E., Schulte, S., Richards, P.L., 2002. Impervious surfaces and water quality: a review of current literature and its implications for watershed planning. Journal of Planning Literature 16 (4), 499–514.

Brumbach, J.J., 1965. The climate of Connecticut. Department of Agriculture and Natural Resources, University of Connecticut.

Carle, M.V., Halpin, P.N., Stow, C.A., 2005. Patterns of watershed urbanization and impacts on water quality. Journal of the American Water Resources Association 41 (3), 693–708.

Coles, J.F., Cuffney, T.F., McMahon, G., and Beaulieu, K.M., 2004. The effects of urbanization on the biological, physical, and chemical characterstics of coastal New England streams. United States Geological Survey, Professional Paper 1695.

Davis, A.P., Shokouhian, M., Sharma, H., Minami, C., 2001. Laboratory study of biological retention for urban stormwater management. Water Environment Research 73 (1), 5–14.

Dietz, M.E., Clausen, J.C., 2005. A field evaluation of rain garden flow and pollutant treatment. Water, Air and Soil Pollution 167 (1–4), 123–138.

Dietz, M.E., Clausen, J.C., 2006. Saturation to improve pollutant retention in a rain garden. Environmental Science and Technology 40 (4), 1335–1340.

Dunne, T., Leopold, L.B., 1978. Water in Environmental Planning. Freeman, San Franciso, CA.

Ferguson, B.K., Suckling, P.W., 1990. Changing rainfall-runoff relationships in the urbanizing Peachtree Creek watershed, Atlanta, Georgia. Water Resources Bulletin 26 (2), 313–322.

Frink, C.R., 1991. Estimating nutrient export to estuaries. Journal of Environmental Quality 20, 717–724.

Gilbert, J.K., Clausen, J.C., 2006. Stormwater runoff quality and quantity from asphalt, paver, and crushed stone driveways in Connecticut. Water Research 40, 826–832.

Groffman, P.M., Law, N.L., Belt, K.T., Band, L.E., Fisher, G.T., 2004. Nitrogen fluxes and retention in urban watershed ecosystems. Ecosystems 7, 393–403.

Hollis, G.E., 1977. Water yield changes after the urbanization of the Canon's Brook catchment, Harlow, England. Hydrological Sciences Bulletin 22, 61–75.

James, L.D., 1965. Using a digital computer to estimate the effects of urban development on flood peaks. Water Resources Research 1 (2), 223–234.

Jennings, D.B., Jarnagin, S.T., 2002. Changes in anthropogenic impervious surfaces, precipitation and daily streamflow discharge: a historical perspective in a mid-Atlantic subwatershed. Landscape Ecology 17, 471–489.

JMP, Version 5 2002. SAS Institute Inc., Cary, NC, 1989–2005.

Klein, R.D., 1979. Urbanization and stream quality impairment. Water Resources Bulletin 15 (4), 948–963.

Leopold, L.B., 1968. Hydrology for urban land planning—a guidebook on the hydrologic effects of urban land use. Geological Survey Circular 554.

Makepeace, D.K., Smith, D.W., Stanley, S.J., 1995. Urban stormwater quality: summary of contaminant data. Critical Reviews in Environmental Science and Technology 25 (2), 93–139.

National Oceanic and Atmospheric Administration, 2006. Climatological Data Summary, Groton, CT, Station #063207. National Climatic Data Center, Asheville, NC.

Novotny, V., Olem, H., 1994. Water Quality: Prevention, Identification, and Management of Diffuse Pollution. Van Nostrand Reinhold, New York.

Pawlow, G.R., Nathan, K., 1977. Impact of suburban development on the rainfall-runoff relationship. Proceedings of International Symposium on Urban Hydrology, Hydraulics, and Sediment Control. University of Kentucky, Lexington, KY July 18–21.

Prince George's County, 1999. Low-impact development design strategies: an integrated design approach. Prince George's County, MD Department of Environmental Resources.

Roy, A.H., Rosemond, A.D., Paul, M.J., Leigh, D.S., Wallace, J.B., 2003. Stream macroinvertibrate response to catchment urbanisation (Georgia, USA). Freshwater Biology 48, 329–346.

Schueler, T.R., 1994. The importance of imperviousness. Watershed Protection Techniques 1 (3), 100–111.

Schueler, T.R., 2003. Impacts of impervious cover on aquatic systems. Watershed Protection Research Monograph No. 1, Center for Watershed Protection, Ellicott City, MD.

Sloto, R.A., 1988. Effects of urbanization on storm-runoff volume and peak discharge of Valley Creek, eastern Chester County, Pennsylvania. Water-Resources Investigation Report 87-4196. United States Geological Survey, Harrisburg, PA.

USDA, 2007. Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture. Official Soil Series Descriptions [Online WWW]. Available URL: 〈http://soils.usda.gov/technical/classification/osd/index.html〉 (accessed 26 January 2007).

US EPA, 1983a. Methods for chemical analysis of water and wastes. United States Environmental Protection Agency, EPA 600/4-79-020. Office of Research and Development, Cincinatti, OH.

US EPA, 1983b. Results of the nationwide urban runoff program. United States Environmental Protection Agency, NTIS PB84-185552.

US EPA, 2000. Low impact development (LID), a literature review. United States Environmental Protection Agency, EPA-841-B-00-005.

US EPA, 2001. Our built and natural environments. United States Environmental Protection Agency, EPA 231-R-01-002.

US EPA, 2002. National water quality inventory, 2000 report. United States Environmental Protection Agency, EPA-841-R-02-001.

Waananen, A.O., 1969. Effects of watershed changes on streamflow. In: Moore, W.L., Morgan, C.W. (Eds.), Urban Effects on Water Yield. University of Texas Press, Austin, London, pp. 169–182.

Wang, L., Lyons, J., Kanehl, P., Bannerman, R.T., 2001. Impacts of urbanization on stream habitat and fish across multiple scales. Environmental Management 28 (2), 255–266.

ELSEVIER

Journal of Environmental Management 87 (2008) 567–581

Journal of
Environmental
Management

www.elsevier.com/locate/jenvman

Multi-scale analysis of oxygen demand trends in an urbanizing Oregon watershed, USA

Mike Boeder, Heejun Chang*

Department of Geography, Portland State University, Portland, OR 97201, USA

Received 20 September 2006; received in revised form 7 November 2007; accepted 1 December 2007

Abstract

Human alteration of the landscape has an extensive influence on the biogeochemical processes that drive oxygen cycling in streams. We estimated trends from the mid-1990s to 2003, using the seasonal Mann–Kendall's test, for percent saturation dissolved oxygen (DO), chemical oxygen demand (COD), total Kjeldahl nitrogen (TKN), and ammonia-nitrogen (NH_3-N) for 12 sites in the Rock Creek watershed, northwest Oregon, USA. In order to understand the influence of landscape change, scale, and stormwater runoff management on dissolved oxygen trends, we calculated land cover change through aerial photo interpretation at full-basin, local (near sample point) basin, and 100 m stream buffer scales, for the years 1994 and 2000. Significant ($p \leqslant 0.05$) trends occurred in DO (increasing at five sites), COD (decreasing at seven sites), TKN (decreasing at five sites, increasing at one site), and NH_3-N (decreasing at one site, increasing at one site). Significant land cover change occurred in agricultural land cover (−8% for the entire basin area) and residential land cover (+10% for the entire basin area) ($p \leqslant 0.05$). Correlation results indicated that: (1) forest cover negatively influenced COD at the full basin scale and positively influences NH_3-N at local scales, (2) residential land cover influenced oxygen demand variables at local scales, (3) agricultural land cover did not influence oxygen demand, (4) local topography negatively influenced TKN and NH_3-N, and (5) stormwater runoff management infrastructure correlated positively with COD at the local scale. This study indicates that landscape factors influencing DO conditions for the study streams act at multiple scales, suggesting that better knowledge of scale-process interactions can guide watershed managers' decision making in order to maintain improving water quality conditions.

Keywords: Water quality; Land cover; Best Management Practices; Scale; Urbanization

1. Introduction

The presence of adequate concentrations of dissolved oxygen (DO) in surface waters is critical to the sustenance of aquatic ecosystems. Low DO concentrations can lead to impaired fish development and maturation, fish mortality, and fish and macroinvertebrate habitat degradation (Rounds and Doyle, 1997; Cox, 2003). The amount of oxygen in a stream is controlled by the volume of water, reaeration by physical processes (e.g., riffles), oxygen production by in-stream plants (biological processes), water temperature, and biochemical oxygen demand (BOD) (the combination of oxygen consumption during the decay of organic matter and the amount of oxygen removed during the nitrification of ammonium). BOD is determined by the amount and type of organic material from point and nonpoint source pollutants as well as from naturally occurring processes. Water temperature relates to oxygen concentration through the solubility of oxygen. With increasing water temperature, oxygen solubility declines, resulting in a reduction in DO (Massoud et al., 2006). Additionally, low flow allows for greater insolation to the water column, increasing water temperature and reducing oxygen saturation concentrations.

In many urban streams, land cover change influences the volume and timing of runoff, which in turn causes multiple water quality impacts, some of which lead to low DO concentrations. In the Portland metropolitan area, watershed urbanization causes the flashiness of

---

*Corresponding author. Tel.: +1 503 725 3162; fax: +1 503 725 3166.
E-mail address: changh@pdx.edu (H. Chang).

0301-4797/$ - see front matter © 2007 Elsevier Ltd. All rights reserved.
doi:10.1016/j.jenvman.2007.12.009

stream discharge as impervious areas (e.g. pavement and compacted soil) interrupt the natural runoff delivery mechanisms of infiltration and groundwater flow (Chang, 2007). This alteration of water movement through the watershed provides a mechanism for increased transport of pollutants and organic materials to receiving waters (e.g. bacteria, leaf litter, sediment, oils and grease), as well as a mechanism for increased stream channel incision and removal of in-stream habitat characteristics such as large woody debris (Dunne and Leopold, 1978; Booth and Jackson, 1997; Wang, 2001; Morley and Karr, 2002; Choi et al., 2003). As urban development increases along Portland's rivers and streams, riparian coverage is reduced, thus altering bank morphology. This results in increased sediment transport, loss of sources for leaf litter and large woody debris, and increased rates of transformation of nutrients (Booth and Jackson, 1997; Band et al., 2005). Finally, the absence of riparian canopy shading in urban stream corridors can warm the water column, facilitating the conversion of adsorbed nutrients to more readily available soluble forms and decreasing in-stream DO values (Karr and Schlosser, 1978).

In order to mitigate negative impacts of urbanization on stream water quality, management agencies in the United States are implementing urban runoff management through Best Management Practices (BMPs). BMPs are structural and non-structural mechanisms employed to control diffuse pollutant loading to receiving waters (Novotny, 2003). Structural BMPs often take the form of areas designed to reduce peak and volume of runoff during a storm event. These areas include retention basins (e.g. ponds, vaults), which capture and attenuate runoff with various flow control structures. Non-structural BMPs include programs such as regularly-scheduled street sweeping to reduce the potential load of leaf litter, sediments, and heavy metals delivered to streams (Tobin and Brinkmann, 2002; Novotny, 2003).

In the Rock Creek basin water quality regulations compel developers to mitigate adverse water quality impacts resulting from urbanization (Oregon Department of Environmental Quality, 2005). As a result of these regulations, an extensive system of storm lines (open and closed conveyances that transport surface runoff), stormwater storage-quality control basins (e.g. drains, vaults, infiltration swales), storm ponds and detention basins has been developed in conjunction with urbanization. Several studies suggest that the connectivity of storm sewer networks and Effective Impervious Area (EIA: impervious surface area that is directly connected to stream channels) is of significant importance to the health of urban streams (e.g. Hatt et al., 2004; McBride and Booth, 2005). Analysis of these stormwater runoff retention-detention basins and other urbanization characteristics such as road density and EIA will provide insight into their influence on oxygen demand at the local scale.

We examined the influence of land cover change and urban storm water management on oxygen demand variables for the Rock Creek basin and its sub-basins using 10 years of data (1994–2003). We employed a multi-scalar approach to spatial analysis to identify scalar linkages with variations in water quality trends. Some studies linked watershed-scale disturbance or sub-basin-scale land cover change with degradation of water quality (e.g., Sliva and Williams, 2001; Morley and Karr, 2002; Chang and Carlson, 2005), while other studies stressed the importance of stream network (Pan et al., 2004). These studies indicates that a multi-scalar approach is appropriate to unravel the complexity of oxygen demand trends in an urbanizing basin.

2. Materials and methods

2.1. Study area

Rock Creek is a tributary of the Tualatin River, adjacent to Portland, Oregon (Fig. 1). The Rock Creek basin encompasses 194 km^2 of the northeastern portion of the Tualatin basin. The headwaters of Rock Creek and its major tributaries are located in the Tualatin Mountains, west and northwest of Portland, at elevations between 200 and 260 m. The mouth of Rock Creek, at its confluence with the Tualatin River at Hillsboro, Oregon, lies at 60 m.

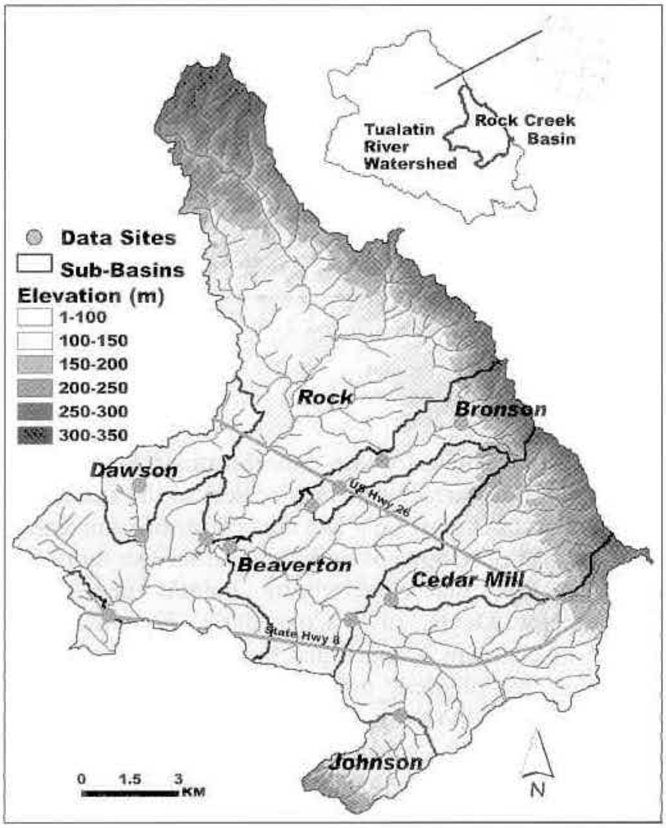


Fig. 1. Rock Creek basin, sub-basins, and trend analysis water quality sites. Hypsographic tints illustrate elevation for the Rock Creek basin. The watershed is crossed by two major arterial roads: US Hwy 26 and Oregon State Hwy 8. Inset maps show the Western US, Tualatin River Basin, and the Rock Creek basin boundary.

We examined 12 monitoring stations that represent a gradient of urban development. Two stations are in the mainstem of Rock creek. Tributary stations include: four in Bronson Creek, two in Beaverton Creek, one in Johnson Creek, one in Cedar Mill Creek (Cedar Mill and Johnson Creeks are tributaries of Beaverton Creek), and two stations in Dawson Creek. A monthly hydrograph for Rock Creek at Quatama Road reflect high flow in wet winter months and low flow in dry summer months (Fig. 2).

The fish habitats of Rock Creek watershed are sensitive to fluctuations in DO and nitrogen in the streams. The mainstem of Rock Creek is a spawning environment for Coho Salmon (*Oncorhynchus kisutch*) and Steelhead Trout (*Oncorhynchus mykiss*). Steelhead trout is listed as threatened under the Endangered Species Act (Oregon Department of Environmental Quality, 2001). Colder headwater reaches of Rock Creek and select tributaries are spawning zones for Cutthroat Trout (*Oncorhynchus clarki*).

Principal municipalities in the Rock Creek basin include Beaverton, Cedar Mill, and Aloha. Portions of Hillsboro extend into the westernmost area of Rock Creek. In 2000, the US Census Bureau (2005) reported the populations of the Aloha and Cedar Mill municipalities to be 41,741 and 12,597, respectively. The 2004 data for Beaverton indicate an increase of 49% from 1990, from 53,307 to 79,350 (City of Beaverton, 2005; Oregon Blue Book, 2005). The Rock Creek basin is traversed east to west by two major transportation arteries, US Hwy 26 and Oregon State Hwy 8. In the 1980s, continuing development pressure in the Tualatin River basin resulted in impaired water quality that could not be mitigated through technological upgrades at point discharge pollutant sources. Impaired oxygen levels and algal blooms were of particular concern. Mitigation measures included Total Maximum Daily Loads (TMDLs), established in compliance with Section 303(d) of the US Clean Water Act (US EPA, 2007), for ammonia (to limit oxygen depletion through nitrification) and phosphorus (to limit algal growth). In 1996 and again in 1998 Rock Creek and its tributaries were listed for impaired dissolved oxygen and temperature. According to Oregon's 2002 section 303(d) list, all tributaries and the mainstem of Rock Creek were de-listed for all water quality parameters. De-listing occurs when monitoring indicates that critical threshold values for respective water quality parameters are being met (Oregon Department of Environmental Quality, 2007).

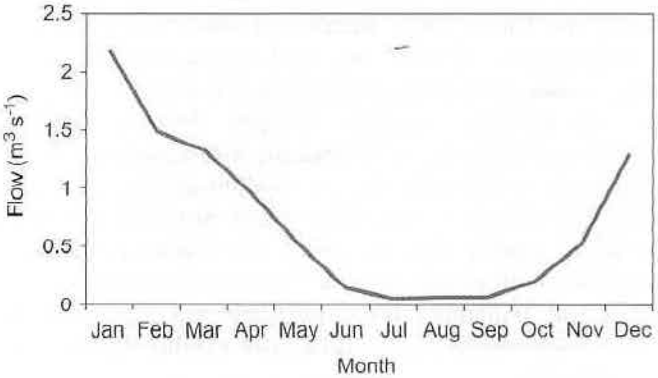


Fig. 2. Monthly hydrograph at Rock Creek at Quatama Road, 2000–2005.

2.2. Data and methods

2.2.1. Water quality and flow data

Water quality data were obtained from Clean Water Services (CWS) of Hillsboro, OR (Clean Water Services, 2005). Samples were typically taken biweekly and analyzed in the CWS laboratory following the standard analytical method for testing water quality (Eaton et al., 1994). Instantaneous flow values were also measured at the time of sample collection. We used four constituents – DO (%sat), COD, TKN, and NH_3-N – for trend analysis at 12 monitoring sites. These sites comprised all of the Rock Creek basin water quality monitoring sites that have long-term instantaneous flow data. All of the flow data records contained gaps. These gaps were addressed in the estimation of trend based upon suggestions by Hirsch et al. (1991) detailed below. In this study, values that fall below the detection limits of laboratory analyses were reported at one-half of the detection limit for the given constituent following the methodology employed in Bekele and McFarland (2004) and Stansfield (2001). Because censored values were treated as ties in the seasonal Kendall test, and the reporting limit fell below actual reported data values (i.e. the reporting limit has not increased through the duration of the data record), censored data had minimal effect on the detection of trend in the current study.

2.2.2. Flow-adjusted concentration: LOWESS residuals

Dissolved chemical concentrations frequently correlate with discharge (Helsel and Hirsch, 1992). In order to account for the influence of discharge variability in trend estimation, Hirsch et al. (1991) and Esterby (1996) recommend measuring trend in residuals from the flow–concentration relationship as determined by Locally Weighted Scatterplot Smoothing (LOWESS). We used this method for our study. Trend estimation using LOWESS residuals has been used in a number of studies, including Zipper et al. (2002) for watersheds in Virginia, US and Djodjic and Bergström (2005) for agricultural watersheds in Sweden.

LOWESS is a robust technique for creating a regression line that is based upon locally weighted averages about each x observation point (Cleveland, 1979). In LOWESS analysis, a window, or smoothing factor, is applied, which identifies the neighborhood of data points around x_0 to be incorporated in the smoothing function. Each weighted average is a function of the magnitude of the residual at point x_0, as well as the distance of x_0 from the center of the moving window width. Smoothing factors range from 0 to 1, with large values minimizing the response of the smoothing function to variability in the data and small values maximizing the response of smoothing to data variability, similar in nature to inverse distance weighting (Bekele and McFarland, 2004). In this study, a smoothing factor of 0.5 (i.e. 50% of the data points incorporated into each smoothing iteration) was chosen. The choice of 0.5 for smoothing is adequate for reducing the variability in constituent concentration attributed to discharge (Bekele and McFarland, 2004). The structure and behavior of the LOWESS function are discussed at length elsewhere (e.g. Cleveland, 1979; Djodjic and Bergström, 2005).

2.2.3. Trend analysis: seasonal Mann–Kendall test

Seasonality can be expressed in a water quality data record and can obscure trend estimation results. Seasonality in water quality parameters can originate from biological and chemical cycling within the watershed as a response to changing hydroclimatic conditions (e.g. timing, intensity, and form of precipitation) that accompany changing seasons. For example, nutrient fluxes in agricultural catchments can rise dramatically at the onset of fall precipitation as ammonium ions adsorbed onto soil particles are flushed off of fields in surface runoff (Heathwaite and Johnes, 1996).

Hirsch and Slack (1984), Gilbert (1987), Hirsch et al. (1991), Helsel and Hirsch (1992), and Esterby (1996) recommend the seasonal Mann–Kendall test as a robust method for accommodating seasonality in trend estimation for water quality records. Numerous researchers have employed this test to estimate trends in water quality data (e.g. Lettenmaier et al., 1991; Yu et al., 1993; Zipper et al., 2002; Räike et al., 2003; Passell et al., 2004). The seasonal Mann–Kendall test computes the nonparametric Mann–Kendall statistic for each user-defined season. The Mann–Kendall test is a modification of the nonparametric Kendall's tau test for correlation, in which data collected over the temporal dimension are correlated with time as the X variable. The Mann–Kendall statistic is computed for each season and the results are combined, removing serial correlation in the data values. Further, estimates of slope or magnitude of monotonic change are derived from the median slope of the ranked slope estimates from the data for each season (Intelligent Decision Technologies, 1998).

2.2.4. Land cover analysis

Multi-scale land cover analysis, utilizing ArcGIS (ESRI, 2005), was accomplished through the establishment of sub-watersheds based on flow accumulation points that corresponded to the water quality monitoring sites (see Table 1). This approach is similar to the approach employed by Morley and Karr (2002) and McBride and Booth (2005). Riparian buffers (100 m) delineate near-stream areas for each stream. This buffer distance is in accordance with Sliva and Williams (2001) and Scott et al. (2002). Multiple local scale watersheds were established within each sub-watershed as well, by locating points along the stream routes, 500 and 1000 m upstream from each monitoring site and using ArcGIS to establish flow accumulation areas (based on the study design of McBride and Booth, 2005). Finally, the 100 m riparian buffer corridors were clipped to local basin boundaries to establish local riparian zones (Fig. 3).

The local analysis portion of this study follows Hatt et al. (2004) and McBride and Booth (2005). Urban land cover variables include road density, storm line density, storm structure density, storm retention structure density (e.g. storm ponds, swales, retention basins, and stormwater vaults), stormwater outfall density, distance to first road crossing, distance to first stormwater outfall, and Effective Impervious Area (EIA). Mean slope of the local basins is also included following the work of Snyder et al. (2003). These variables are assessed for each 1000 m basin derived in the previous analysis. The EIA data set is based upon an EIA assessment for the year 2000 (Clean Water Services, 2005).

For land use classification, we used 1994 aerial photographs (georectified root-mean-squared errors less than 10 m) and 2000 aerial imagery (a georeferenced photomosaic established by the US Geological Survey). Wetland coverage data from 1998 was based upon locally revised US National Wetlands Inventory data. We used this supplementary data set to locate wetlands given the difficulty of wetland identification through visual interpretation of aerial photography (see Appendix 1). Land use classifications follow a modified Anderson Level II classification (Anderson et al., 1976) (see Appendix 2). This classification system has been used for categorizing land cover based on aerial photo imagery (e.g. urban or built-up land in Anderson Level I can be subdivided into residential, commercial, industrial, transportation, etc.) (Anderson et al., 1976). Land cover polygons representing different land covers were then converted to a 30 m raster grid, which allowed for the estimation of land cover change using simple grid addition.

2.2.5. Correlation and multiple comparisons

We used correlation analysis to determine the relationship between land cover and water quality. Correlation analysis has been used extensively in other similar studies (Gove et al., 2001; Stewart et al., 2001; Scott et al., 2002; Tong and Chen, 2002). Because of small sample size and non-normality of data, we used non-parametric Spearman's rank correlation coefficients for all land cover and oxygen demand variables. Oxygen demand data are disaggregated by season to coincide with seasonal divisions employed in the trend analysis portion of this study (dry: June–October; wet: November–May). Seasonal separation of water quality data in land cover studies is based on monthly hydrograph (see Fig. 2) and follows the work of Sliva and Williams (2001), and their assertion that land cover/water quality correlations can exhibit strong seasonal response.

When comparing multiple correlations between water quality and different land covers, we used the false

Table 1
Seasonal Kendall test results

Site name	Data record	DO (% sat)		COD (mg/L)		TKN (mg/L)		NH_3-N (mg/L)	
		N	Slope[b]	*N*	Slope[b]	*N*	Slope[b]	*N*	Slope[b]
Bronson at Saltzman	8/97–9/03	132		138		129		138	
Bronson at West Union	6/95–9/03	174		180	−0.442	172	−0.009	157	0.001
Bronson at Bronson Park	2/01–9/03	56		57		50	−0.036	58	
Bronson at 185th	5/94–2/97	76		75		75		NA	
Beaverton at 170th[a]	5/96–8/03	79	1.980	80	−0.767	77	−0.012	80	
Beaverton at Cornelius Pass[a]	5/90–10/00[c]	83	1.143	78	−0.739	84		31	
Cedar Mill at Jenkins	6/96–4/01	59	3.314	62	−1.290	62		61	
Dawson at Airport	4/01–4/03	35		35		30		35	
Dawson at Brookwood	7/97–9/03	116		117		109		117	
Johnson at Davis	5/94–9/03[d]	176	2.384	178	−0.435	171	0.010	86	
Rock Creek at Quatama A	5/91–11/95	136		123		135		NA	
Rock Creek at Quatama B	7/98–9/03	164		167	−0.680	152	−0.017	167	
Rock Creek at Hwy 8[a]	5/90–3/03[e]	130	0.513	124	−0.814	128	−0.005	75	−0.001

Data indicate trend in LOWESS residuals from the flow–concentration relationship for water quality sites in the Rock Creek basin.
Trend results significant at 95% confidence level ($\alpha = 0.05$) are only reported here.
NA = insufficient observations.
[a] Trend analysis for these three sites was computed on monthly median values because of computational limitations of the software.
[b] Slope values are based on Seasonal Kendall Sen Slope estimator in units/yr.
[c] 5/96–10/00 for NH_3-N.
[d] 5/99–9/03 for NH_3-N.
[e] 6/91–3/03 for COD and 5/96–3/03 for NH_3-N.

discovery rate (FDR) to remove the harshness of standard procedures such as Bonferroni corrections (which control the risk of making at least one Type I error), by controlling the permissible proportion of Type I errors. FDR is defined as "the expected proportion of true hypothesis rejected out of the total number of rejections (McBride, 2005)." The cutoff *p*-value is determined if the *p*-value is less than the ratio of the *N*th lowest *p*-value to the number of comparisons times the false discovery rate (e.g., $\alpha = 0.05$). FDR is advantageous over Bonferroni correction because it regains statistical power and avoids an abundance of false-positive results. The FDR has been used in previous water quality studies (e.g., Garcia, 2003; Niyogi et al., 2007).

3. Results

3.1. Trend analysis

Table 1 illustrates the results for the seasonal Mann–Kendall test of the LOWESS residuals from the flow–concentration relationship. Significant increasing trends were found for DO (%sat) at the Beaverton Creek sites, Cedar Mill Creek, Johnson Creek, and Rock Creek at Hwy 8 (range: 0.513–3.314%/yr). Significant decreasing trends in COD were indicated for Bronson Creek at West Union, the Beaverton Creek sites, Cedar Mill Creek, Johnson Creek, late Rock Creek at Quatama, and Rock Creek at Hwy 8 (range: −0.442 to −1.290 mg/L/yr). Significant decreasing trends in TKN were found at Bronson Creek at

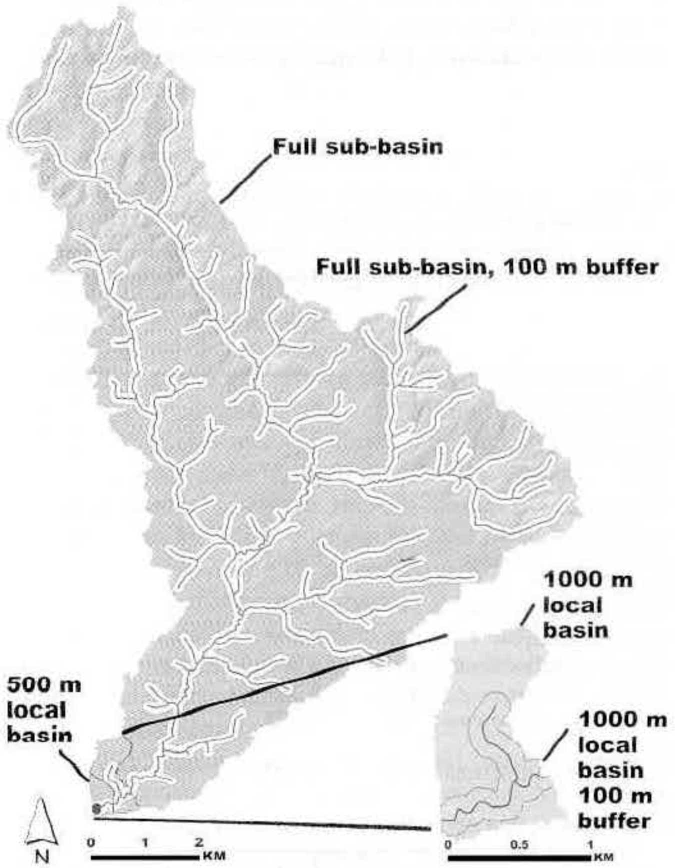


Fig. 3. Boundary delineation for multi-scale land cover assessment. The enlarged inset represents a 1000 m sub-basin delineated from the water quality sampling point. Buffer distance in both figures is 100 m.

West Union and Bronson Park, Beaverton Creek at 170th, Rock Creek late Quatama, and Rock Creek at Hwy 8 (range: −0.005 to −0.036 mg/L/yr). An increasing trend in TKN was reported for Johnson Creek (0.010 mg/L/yr). Finally, NH_3-N data showed significant trends at Bronson Creek at West Union (0.001 mg/L/yr) and Rock Creek at Hwy 8 (−0.001 mg/L/yr).

Table 2 shows the results for Spearman rank correlation between nitrogenous oxygen demand variables and COD. At all stations, COD varies significantly with TKN (correlation values from 0.26 at the Johnson Creek site to 0.63 at the Cedar Mill Creek site). Dissolved NH_3-N only varies significantly with COD at two stations, the headwaters Bronson Creek at Saltzman, and Dawson Creek at Brookwood (0.23 and 0.19, respectively).

A complicated picture emerges when the values from the correlation analysis are applied to trend direction. At the Bronson Creek at West Union site, both TKN and COD exhibit decreasing trends and show a moderately strong correlation between TKN and COD (Spearman's rank correlation coefficient $\rho = 0.61$). The Beaverton Creek at 170th avenue site also exhibits decreasing trends in TKN and COD, but there is only a moderate correlation between COD and TKN ($\rho = 0.48$). The Rock Creek sites (Quatama Road late and Hwy 8) exhibit decreasing trends in TKN and COD with ρ values of 0.36 and 0.39, respectively. Additionally, NH_3 concentrations exhibit a downward trend, albeit small in magnitude (−0.001), at the Rock Creek at Hwy 8 site.

3.2. *Land cover analysis*

Percent land cover values were determined for each assessment scale (full basin, full basin 100 m stream buffer, local basin (500, 1000 m), and local basin 100 m stream buffers (see Appendix 3). Aggregated data for all assessment scales was analyzed for statistical difference based on groupings by year using the Wilcoxon Signed Ranks test. Results (not shown) indicate that aggregate values for agriculture and residential land cover designations are statistically distinct between 1994 and 2000 at $p \leqslant 0.001$ ($N = 72$). Basin-wide results for land cover change indicate that there was an 8% loss of agricultural area (1542 ha) and a 10% increase in residential area (1873 ha). Fig. 4 illustrates conversion of land to residential and commercial land uses from 1994 to 2000.

Tables 3 and 4 summarize the correlations between land cover and seasonally disaggregated (dry and wet season) oxygen demand variables for the mid-1990s and 2000, respectively (confidence level: 95% ($p \leqslant 0.05$)). As shown in these tables, significant correlations are more pronounced at the local basin and buffer scales than full sub-basin and buffer scales. At the full sub-basin scale and the full buffer scale, only eight significant correlations exist. Dry season COD consistently shows negative correlation with forest land cover at both scales for both years. For mid-1990s wet season DO is negatively associated with commercial land cover, while dry season DO is positively associated with wetland land cover at both scales. No significant correlations are found for the 2000 wet season data.

At the 1000 m basin scale, no consistent results are found in either years. For mid-1990s data, NH_3-N is negatively associated with open land in both seasons, while dry season DO is positively associated with residential land cover. Significant negative correlations exist between dry season TKN and roads and between NH_3-N and water. For the 2000 data, dry season NH_3-N is positively correlated with agricultural land while it is negatively associated with roads and residential land cover. At the 1000 m buffer scale, forest land cover is positively associated with mid 1990s dry season NH_3-N concentration and wet season NH_3-N concentration. Dry season NH_3-N is negatively associated with roads and residential land cover. At the 500 m buffer scale, dry season NH_3-N is positively associated with forest land cover for both seasons in 2000, but only significant for dry season in the mid-1990s. Dry season DO is negatively associated with water and residential areas in the mid-1990s and with forest in 2000.

The FDR analysis drastically reduces the number of statistically significant correlations in our data set. The cutoff p-value is 0.00625. Only two cases meet this criterion: negative correlation between dry season NH_3-N concentration and open land at the 1000 m basin scale ($p = 0.003$), and dry season DO saturation and residential land at the 1000 m basin, 100 m buffer scale ($p = 0.001$). As will be discussed later, reducing the discussion to only those

Table 2
Spearman rank correlation results for relationships between TKN, NH_3-N and COD at Rock Creek basin study sites

Site name	COD and TKN		COD and NH_3-N	
	N	Spearman coeff.	*N*	Spearman coeff.
Bronson at Saltzman	129	0.38**	138	0.23**
Bronson at West Union	172	0.61**	157	0.06
Bronson at Bronson Park	58	0.35**	65	−0.14
Bronson at 185th	75	0.49**	Insufficient data	
Beaverton at 170th[a]	169	0.48**	145	−0.03
Beaverton at Cornelius Pass[a]	225	0.43**	85	0.12
Cedar Mill at Jenkins	63	0.63**	62	0.03
Dawson at Airport	30	0.38**	35	0.16
Dawson at Brookwood	109	0.34**	117	0.19*
Johnson at Davis	171	0.26**	106	0.18
Rock Creek at Quatama (early)	122	0.47**	Insufficient data	
Rock Creek at Quatama (late)	152	0.36**	167	0.06
Rock Creek at Hwy 8[a]	416	0.39**	216	0.04

*Results significant at 95% confidence level ($\alpha \leqslant 0.05$). **Results significant at 99% confidence level ($\alpha \leqslant 0.01$).

[a] Trend analysis for these three sites was computed on monthly median values because of computational limitations of the software.

correlations that are statistically significant runs the risk of dismissing ecologically significant relationships that don't meet an established p-value. With this in mind, and accepting the potential for Type I error, we will discuss the results of the uncorrected correlation analysis. That is, we feel that the uncorrected correlation analysis results suggest important relationships between water quality and land covers that may be unnecessarily dismissed under the guidance of FDR results.

Table 5 contains Spearman's correlation results for 1000 m local basin analysis between basin urban stormwater management infrastructures and median oxygen demand values for seasonal and annually aggregated data from 2000. The 2000 dry season data shows two significant ($p \leqslant 0.05$) correlation values: mean slope correlates negatively with TKN and NH_3-N. The wet season data also exhibits a significant negative correlation between mean slope and NH_3-N. Additionally, COD correlates positively with storm line density, storm structure density, storm retention structure density, and storm outfall density. Even though multiple comparisons are considered, these are all significant. Storm outfall density also exhibits significant positive correlation with TKN.

4. Discussion

4.1. Trend analysis

The results of trend analysis are dependent on the period studied and sample size. In a previous study in the Bronson Creek watershed, Creech (2003) found trends of decreasing total nitrogen from 1994 to 2001 at nine sites, significant declining trends in NH_3-N for four upstream Bronson Creek sites, and one increasing trend for the lowest Bronson Creek site. The results were based on the Mann–Kendall test on seasonally disaggregated non-flow-adjusted water quality values. In the present study, data for total nitrogen over a similar period showed significant ($p \leqslant 0.05$) decreasing trends for two sites and one significant increasing trend in NH_3-N (0.001 mg/L/yr, $p \leqslant 0.05$). Potential explanations for these differences include the use of flow correction in the present study, and differences in sample numbers between the two studies.

As a cautionary note, a single p-value may not capture the potential environmental consequences of landscape change. In other words, statistical insignificance (where p-value is greater than 0.05) does not necessarily mean that trends are environmentally insignificant (McBride, 2005). A small insignificant difference could have implications for larger environmental change.

The correlation analysis indicates that nitrogenous BOD plays an important role in oxygen dynamics for many of the sites tested. Owing to high levels of bioavailable orthophosphate in this watershed (Wilson et al., 1999), it may be that nitrogen bound in phytoplankton biomass explains the majority of organically bound nitrogen. Further, these results demonstrate that oxygen demand management in the Rock Creek basin should address nitrogenous inputs and conditions that may influence nitrogenous oxygen demand.

4.2. Land cover analysis

Land cover analysis results for this study demonstrate that oxygen demand variables are influenced to different degrees by land cover classes, depending on the scale of land cover assessment. The negative correlations between forest land cover and oxygen demand variables at the sub-basin-wide and the sub-basin stream buffer scales suggest the importance forest cover throughout the entire sub-basin in mitigating the delivery of oxygen demanding materials to receiving waters, most likely through the retention of decaying organic matter. While there was approximately 5% (958 ha) of forest land loss in the Rock Creek basin north of Hwy 8 between 1994 and 2000, this loss is insufficient to alter the sub-basin-wide influence of forest cover on COD concentrations. Sub-basin scale forest cover assessments do not exhibit correlation with nitrogenous variables. Hence, these results indicate that forest land cover influences the carbonaceous component of biochemical oxygen demand (otherwise, ammonium and TKN correlations would mirror the negative relationship demonstrated by COD).

The positive correlation between ammonium and forest cover at the local scale indicates that ammonia export via pathways, such as leaching and throughfall, may exceed uptake by flora. It is possible that dense riparian vegetation adds substantial organic matter to the channel. Organic matter decomposition could release ammoniacal-nitrogen to water bodies. Similarly, Scott et al. (2002), in a study of several forested Tennessee River basins, found that strong positive correlation ($r^2 = 0.66$) between ammonium and forest land cover at a 100 m buffer scale. These results emphasize for watershed management the importance of maintaining forest cover in order to mitigate carbonaceous oxygen demand in surface waters from both a sub-basin-wide perspective and a sub-basin, near stream perspective.

The disappearance of the negative correlation between commercial land cover and wet season DO (%sat) values in 2000 at the sub-basin scale suggests that the role of urban runoff management measures accompanying new development (3% increase in commercial land) is important in improving DO conditions. However, local basin analysis of urban runoff management refutes this suggestion. Wet season median COD data for 2000 demonstrate significant positive correlations with urban runoff management variables. A more detailed local analysis might reveal a linkage between urban landscape variables and COD.

Residential land cover exerts more influence over oxygen demand in the Rock Creek basin at local scales than sub-basin scales. As residential land cover increased (10% increases between 1994 and 2000), the positive influence of this land cover class on in-stream DO declined. These results indicate that watershed management decisions

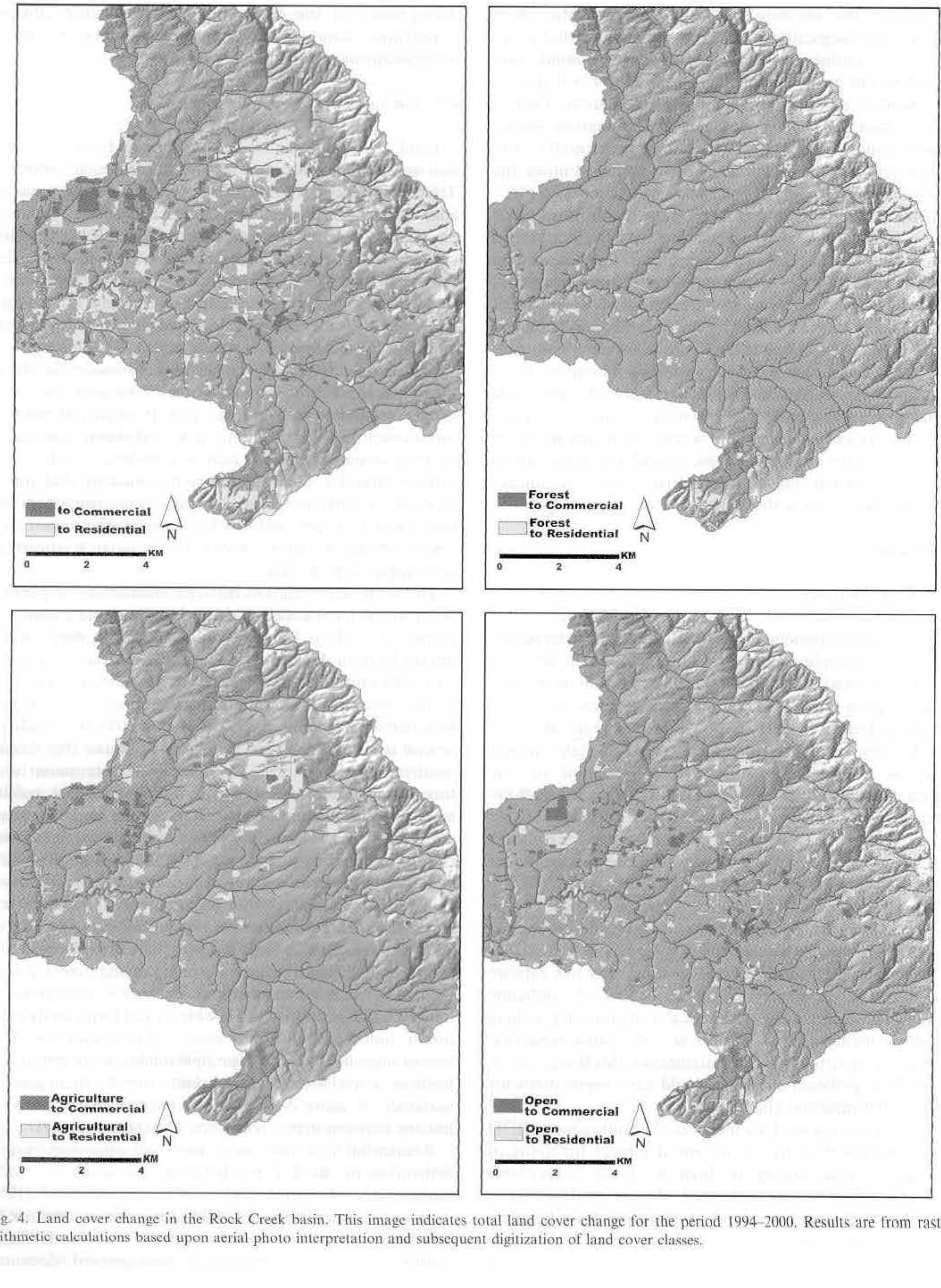


Fig. 4. Land cover change in the Rock Creek basin. This image indicates total land cover change for the period 1994–2000. Results are from raster arithmetic calculations based upon aerial photo interpretation and subsequent digitization of land cover classes.

Table 3
Mid-1990s correlation results for land cover and oxygen demand variables

			Agri	Com	Forest	Roads	Water	Wetland	Open	Resident
Full sub-basin										
Dry	DO (%sat)	ρ						0.661		
		p-value						0.019		
		N						12		
	COD	ρ			−0.694					
		p-value			0.012					
		N			12					
Wet	DO (%sat)	ρ		−0.582						
		p-value		0.047						
		N		12						
1000 m basin										
Dry	DO (%sat)	ρ								0.638
		p-value								0.025
		N								12
	TKN	ρ				−0.580				
		p-value				0.048				
		N				12				
	NH_3-N	ρ					−0.787		−0.919	
		p-value					0.036		0.003*	
		N					7		7	
Wet	NH_3-N	ρ							−0.821	
		p-value							0.023	
		N							7	
500 m basin										
Dry	DO (%sat)	ρ					−0.656			0.603
		p-value					0.020			0.038
		N					12			12
Wet	DO (%sat)	ρ								0.677
		p-value								0.016
		N								12
Full-Buffer										
Dry	DO (%sat)	ρ						0.714		
		p-value						0.009		
		N						12		
	COD	ρ			−0.658					
		p-value			0.020					
		N			12					
Wet	DO (%sat)	ρ		−0.649						
		p-value		0.023						
		N		12						
1000 m-buffer										
Dry	DO (%sat)	ρ								0.818
		p-value								0.001*
		N								12
	NH_3-N	ρ			0.847		−0.787			
		p-value			0.016		0.036			
		N			7		7			
500 m-buffer										
Dry	DO (%sat)	ρ					−0.656			0.734
		p-value					0.020			0.007
		N					12			12
	TKN	ρ		−0.728						
		p-value		0.007						
		N		12						
	NH_3-N	ρ			0.847				−0.811	
		p-value			0.016				0.027	
		N			7				7	

Data reflect Spearman's correlation values for land cover data assessed at multiple scales and seasonal median oxygen demand values. Only significant relationships are reported here ($p \leq 0.05$); *indicates the cutoff p-value using the false discovery rate $\alpha = 0.05$.

Table 4
The 2000 correlation results for land cover and oxygen demand variables

			Agri	Com	Forest	Roads	Water	Wetland	Open	Resident
Full sub-basin										
Dry	COD	ρ			−0.620					
		p-value			0.042					
		N			11					
1000 m basin										
Dry	NH_3-N	ρ	0.616			−0.634				−0.620
		p-value	0.044			0.036				0.042
		N	11			11				11
Wet	COD	ρ			0.588					
		p-value			0.044					
		N			12					
	NH_3-N	ρ					−0.610			
		p-value					0.035			
		N					12			
Full-buffer										
Dry	COD	ρ			−0.756					
		p-value			0.007					
		N			11					
1000 m-buffer										
Dry	NH_3-N	ρ				−0.726				−0.688
		p-value				0.011				0.019
		N				11				11
Wet	NH_3-N	ρ			0.592		−0.610			
		p-value			0.043		0.035			
		N			12		12			
500 m-buffer										
Dry	DO (%sat)	ρ			−0.697					
		p-value			0.025					
		N			10					
	NH_3-N	ρ			0.758					
		p-value			0.011					
		N			10					
Wet	NH_3-N	ρ			0.642					
		p-value			0.033					
		N			11					

Data reflect Spearman's correlation values for land cover data assessed at multiple scales and seasonal median oxygen demand values. Only significant relationships are reported here ($p \leqslant 0.05$); *indicates the cutoff p-value using the false discovery rate $\alpha = 0.05$.

regarding oxygen demand in surface waters must address the local influence of residential land cover. Residential and commercial development in the Rock Creek basin implies a change in runoff dynamics for the watershed as the natural infiltration-to-baseflow process is interrupted by increasing impervious surfaces (Chang, 2007). Local assessment of urban runoff management variables provides further insight into the landscape/oxygen demand relationship in this basin.

Connectivity, in reference to urban runoff pathways, describes the direct linkages of impermeable surfaces to receiving waters, typically via stormwater conveyance structures. Hatt et al. (2004), in a study of fifteen streams near Melbourne, Australia, found that ammonium correlated strongly ($r^2 = 0.71$) to drainage connection. They cited connectivity as a more sensitive indicator than total impervious surface area for water quality constituent concentrations (e.g. nutrients, suspended solids, dissolved organic carbon). The Rock Creek results do not support this finding. In the Rock Creek data, NH_3-N values are not correlated to any of the connectivity indicators (stormwater line density, distance to road crossing, distance to stormwater line outfall, EIA), demonstrating a complex relationship that is not adequately captured by simple road connectivity measures. With increasing population density, stormwater management facilities may not be adequately mitigating the delivery of oxygen demanding materials to surface waters. The strong positive correlation between stormwater management infrastructure variables and wet season median COD concentrations suggest that population density as reflected by the number of stormwater infrastructures in general is driving COD trends.

Table 5
Spearman's correlation results between urban runoff management variables and seasonal median oxygen demand data for 2000

			Mean slope	Road density	Storm line density	Storm structure density	Storm retention strure density	Storm outfall density	Distance to Road crossing	EIA
Dry	TKN	ρ	−0.694							
		p-value	0.018							
		N	11							
	NH_3-N	ρ	−0.679							
		p-value	0.022							
		N	11							
Wet	COD	ρ			0.819	0.654	0.805	0.779		
		p-value			0.001*	0.021*	0.002*	0.003*		
		N			12	12	12	12		
	TKN	ρ						0.579		
		p-value						0.048		
		N						12		
	NH_3-N	ρ	−0.641							
		p-value	0.025							
		N	12							

*Indicates the cutoff p-value using the false discovery rate $\alpha = 0.05$.

Finally, local topography exhibits negative correlations with dry season TKN data and wet season NH_3-N data (Table 5). This result supports the findings of Snyder et al. (2003), which indicate that steeper slopes in urban areas of a West Virginia watershed had more influence over stream health indices. In the present study, steeper local watershed gradients are primarily associated with the Bronson Creek at Saltzman Road site, where forest cover dominates the landscape. Previous discussion described the variability found in the relationship between forest cover and oxygen demand in the Rock Creek basin. Further analysis is required to adequately explain the influence of slope at the local scale over oxygen demand.

5. Conclusions

Trend analysis results indicate that DO conditions in streams are improving throughout the Rock Creek basin with multiple sites reporting increases in DO (%sat) (0.513–3.314%/yr) and decreases in COD (−0.442 to −1.290 mg/L/yr), TKN (−0.005 to −0.036 mg/L/yr), and NH_3-N (−0.001 mg/L/yr). In order to explore potential linkages between land cover change and water quality trends in the Rock Creek basin, a land cover change assessment was completed at the sub-basin, stream buffer, local basin (500 and 1000 m drainage basins), and local basin buffer scales based on visual interpretation of aerial imagery from 1994 and 2000. Significant ($p \leqslant 0.001$) land cover change over this time period occurred in agricultural land cover (−8% for the entire basin) and residential land cover (+10% for the entire basin). Correlation analysis established numerous statistically significant relationships between seasonally disaggregated median oxygen demand variables and land cover classifications for the mid-1990s—2000. These results support the importance of scale in identifying land cover/water quality relationships. Forest cover was found to influence the mitigation of COD levels for surface waters at the full basin scale and full basin stream buffer scale. Local scale basin and near-stream buffer area analysis indicated that residential land cover positively influenced stream DO (%sat) values during the mid-1990s. This relationship was not present in the 2000 data. Near-stream forest cover correlated positively with dry season NH_3-N values for the mid-1990s and wet season NH_3-N values for 2000. Hence, while full basin forest cover mitigates the delivery of carbonaceous oxygen demanding materials to streams, local influences control the relationship between forest cover and nitrogenous variables.

The suggestion that local mechanisms are important in determining oxygen demand conditions for Rock Creek and its tributaries encouraged a more detailed analysis of urban runoff management variables and seasonally disaggregated oxygen demand data for 2000. Contrary to trends in improving oxygen demand characteristics for Rock Creek streams, urban runoff management variables correlated positively with COD during the wet season. Connectivity metrics as well as EIA did not produce significant correlations with oxygen demand variables. Results from this study demonstrate that watershed management must account for mechanisms that influence oxygen demand at varying scales.

Rock Creek basin streams were de-listed for DO and temperature on the 2002 Oregon 303d list. While removal criteria are based on the maintenance of critical thresholds and not long-term trends, trend results from this study support the de-listing of these streams (Oregon Department of Environmental Quality, 2005). The question remains, however, whether trends will continue to approach their physical maxima or minima (i.e. physical limits for these oxygen parameters) or whether trends in oxygen demand variables will reverse as development pressures overwhelm the built and natural mechanisms for oxygen demand mitigation. This study provides a basis from which to more accurately examine and manage the complex mechanisms that drive oxygen demand in urbanizing streams.

Acknowledgements

This research was supported by a Faculty Enhancement Grant at Portland State University. Additional support was provided by Department of Geography Perry Award.

We appreciate Jill Ory and Jan Miller at Clean Water Services for providing water quality data. Three anonymous reviewers' suggestions greatly helped improve the quality of the manuscript. Jon Franczyk at Portland State University also helped clarify some points in the manuscript.

Appendix 1

Data sources and resolution for spatial data sets used in land cover change analysis and local urban land cover, urban runoff management analysis.

Data type	Source	Resolution
Digital elevation model	USGS Seamless	National elevation data set 10 m
Aerial photography 9 July 1994	University of Oregon Map & Aerial Photography Library: Northern Lights Project	1″ = 2000 Photos scanned at 600 dpi
Aerial photography 2000	USGS Seamless	Digital Orthoimagery Quarter Quadrangle 1 m
Wetland data set 1998	Metro Regional Land Information System (RLIS) 2005	National Wetlands Inventory with local revisions carried out by Tri-county government agencies; 40 ft positional accuracy
Stream route shapefile	Metro Regional Land Information System (RLIS) 2005	Stream lines derived from a variety of sources.
Watershed boundary shapefiles	Metro Regional Land Information System (RLIS) 2005	6th field HUCs
Stormwater management system shapefiles	Clean Water Services 2004	Washington County stormwater structure and drainage line database.

Appendix 2

Land cover classifications used in aerial photo interpretation. Land cover classes are based upon modified Anderson Level II land cover classes (Anderson et al., 1976).

Land cover classification	Description
Agriculture	Land surface that shows evidence of obvious cultivation through parallel lines in field surfaces (e.g. crop rows) including vineyards and orchards. Includes buildings on agricultural land (barns, farmhouses, etc.)
Open	Open canopy, not cultivated, but shows signs of use. Includes parks, playing fields, golf courses, etc. Also includes apparently unused land, construction areas and logged areas.
Major roads	Roads four lanes or wider, or divided. Includes medians and associated open space (e.g. within cloverleafs)
Residential	Residential neighborhoods, including streets, medians, and schools.
Commercial	Large buildings that are obviously commercial in nature. Apartment buildings and schools are excluded, as they generally appear in residential neighborhoods. Includes parking lots and minor open spaces between buildings.
Forest	Closed canopy that is more extensive than neighborhood trees. Includes closed canopy over parks and stream channels.
Open water	Lakes, ponds, stock ponds.

Appendix 3

Land cover changes for 12 monitoring sites at the full basin, 1000 m basin, and 500 m basin, and respective 100 m buffer riparian scales.

	1990s								2000							
	Agri	Com	Forst	Road	Watr	Wet	Open	Res	Agri	Com	Forst	Road	Watr	Wet	Open	Res
Full basin																
BrnSalt	9.2	0.0	72.5	0.0	0.0	0.0	12.0	6.3	4.0	0.0	68.0	0.0	0.0	0.0	17.2	10.7
BrnWU	30.8	0.2	47.2	0.0	0.1	0.6	11.3	9.7	14.7	0.6	44.6	0.0	0.0	0.6	17.3	22.2
BrnBP	27.4	0.3	39.0	0.6	0.1	1.6	11.6	19.4	12.6	0.7	37.0	0.7	0.0	1.6	15.9	31.6
Bronson	24.8	1.3	35.3	1.3	0.1	1.9	14.0	21.3	11.4	3.5	33.6	1.6	0.0	1.9	15.3	32.5
Bvtn170	1.9	11.7	19.1	2.2	0.0	1.3	12.1	51.6	1.5	11.4	16.5	2.7	0.0	1.2	8.8	57.8
BvtnCP	6.6	9.8	18.1	1.9	0.0	1.3	13.5	48.8	3.1	10.4	16.1	2.4	0.0	1.2	11.3	55.4
DawAir	44.1	6.0	3.5	4.7	0.3	0.0	31.4	10.0	14.9	19.6	2.2	3.4	0.3	0.0	32.2	27.5
Dawson	37.3	7.1	6.5	7.3	0.2	0.8	28.4	12.4	15.0	17.1	4.8	3.7	0.2	0.8	33.5	25.0
Cedar mill	3.7	6.0	30.4	2.5	0.1	0.7	12.3	44.3	2.6	6.6	21.5	2.9	0.1	0.7	9.5	56.1
Johnson	2.4	0.6	29.5	0.6	0.0	2.0	16.4	48.5	2.7	0.9	22.7	0.8	0.0	2.0	7.6	63.3
Quatama	37.4	1.5	35.2	0.6	0.1	1.0	17.4	6.9	30.4	3.6	37.3	0.7	0.1	1.0	14.9	11.9
RCHwy8	19.5	6.4	22.8	1.6	0.0	1.1	15.8	32.7	13.4	8.3	22.5	1.6	0.0	1.0	14.2	39.0
1000 m Basin																
BrnSalt	11.5	0.0	69.2	0.0	0.0	0.0	11.5	7.8	4.0	0.0	66.1	0.0	0.0	0.0	18.7	11.3
BrnWU	44.4	0.0	2.5	0.0	0.7	1.0	15.5	36.0	0.0	1.5	6.5	0.2	0.0	1.0	17.6	73.2
BrnBP	17.3	0.8	1.8	5.4	0.0	6.0	14.9	53.9	4.7	1.6	2.6	5.6	0.0	6.0	11.2	68.3
Bronson	0.0	9.5	0.0	3.8	0.3	4.5	36.9	44.9	0.0	31.1	1.3	6.5	0.3	4.5	7.5	48.8
Bvtn170	0.0	14.4	45.5	0.0	0.0	11.4	11.5	17.3	0.0	13.3	48.5	0.1	0.0	11.1	13.1	13.9
BvtnCP	17.9	0.0	9.4	0.0	0.0	0.0	24.9	47.8	14.7	0.0	8.3	0.0	0.0	0.0	29.4	47.5
DawAir	35.6	5.8	6.7	2.7	0.8	0.0	39.6	8.8	19.9	12.1	3.7	0.0	0.5	0.0	34.7	29.0
Dawson	26.3	8.6	18.7	1.3	0.0	1.9	12.2	31.0	14.5	12.2	12.0	1.4	0.0	1.9	22.2	35.8
Cedar mill	0.0	23.1	10.4	1.4	0.1	2.2	38.0	24.7	0.3	35.4	11.6	3.1	0.0	2.2	19.7	27.6
Johnson	0.0	0.3	16.9	2.5	0.0	3.9	5.2	71.2	0.0	1.1	15.7	2.7	0.0	3.9	4.6	71.9
Quatama	33.0	1.7	21.5	0.0	0.0	0.0	33.1	10.7	20.9	10.1	11.5	0.0	0.0	0.0	38.8	18.8
RCHwy8	5.5	12.8	19.6	3.8	0.0	6.6	38.5	13.3	1.1	18.5	14.3	0.0	0.0	6.3	37.5	22.3
500 m Basin																
BrnSalt	14.5	0.0	70.0	0.0	0.0	0.0	13.9	1.6	6.5	0.0	64.8	0.0	0.0	0.0	25.9	2.7
BrnWU	46.3	0.0	4.2	0.0	0.8	2.9	22.4	23.4	0.0	0.0	11.9	0.5	0.0	2.9	24.5	60.2
BrnBP	17.7	1.2	1.6	8.7	0.0	4.7	19.6	46.4	7.6	2.6	3.5	9.0	0.0	4.8	11.0	61.4
Bronson	0.0	9.3	0.0	3.5	0.0	4.7	35.6	46.9	0.0	28.0	1.3	5.7	0.0	4.7	4.9	55.3
Bvtn170	0.0	25.1	39.2	0.0	0.0	7.0	20.8	0.0	0.0	11.7	43.2	0.2	0.0	7.7	9.5	27.8
BvtnCP	15.3	0.0	5.7	0.0	0.0	26.6	52.4	0.0	13.0	0.0	7.0	0.0	0.0	0.0	34.9	45.1
DawAir	62.4	0.0	10.6	4.6	2.2	0.0	18.0	2.2	28.3	17.2	9.5	0.0	1.0	0.0	34.4	9.6
Dawson	15.5	0.1	17.1	0.0	0.0	1.1	13.0	53.2	3.0	3.3	12.4	0.0	0.0	1.1	21.6	58.7
Cedar mill	0.0	26.4	9.9	1.1	0.0	1.9	39.0	21.7	0.4	40.4	9.8	3.4	0.0	1.9	19.2	24.8
Johnson	0.0	0.4	16.5	3.0	0.0	1.8	2.2	76.1	0.0	0.7	15.4	3.2	0.0	1.8	1.2	77.7
Quatama	0.0	0.0	29.4	0.0	0.0	0.0	37.0	33.6	0.0	3.0	16.7	0.0	0.0	0.0	37.8	42.5
RCHwy8	3.0	21.2	18.0	6.4	0.0	36.3	8.3	0.0	0.0	22.4	19.2	0.0	0.0	6.7	29.8	21.9
Full rip																
BrnSalt	6.2	0.0	83.7	0.0	0.0	0.0	7.2	2.9	3.2	0.0	83.7	0.0	0.0	0.0	8.3	4.8
BrnWU	23.6	0.0	59.2	0.0	0.1	1.6	11.4	3.9	9.7	0.0	59.7	0.1	0.0	1.6	17.8	11.2
BrnBP	20.7	0.0	51.3	0.4	0.1	5.3	13.1	9.0	8.4	0.0	52.0	0.5	0.0	5.3	18.0	15.7
Bronson	19.2	0.3	47.4	1.0	0.2	6.5	15.6	9.9	7.8	0.2	48.2	1.4	0.0	6.5	18.6	17.2
Bvtn170	1.2	9.3	24.5	1.8	0.0	3.6	13.8	45.8	0.8	9.3	23.7	2.2	0.0	3.4	10.5	50.0
BvtnCP	4.8	7.6	24.5	1.8	0.0	3.9	15.9	41.4	2.0	8.3	24.1	2.2	0.0	3.7	14.4	45.3

DawAir	46.7	2.8	6.7	6.0	1.1	0.0	34.9	1.7	17.6	6.7	5.5	4.3	0.9	0.0	49.2	15.9
Dawson	37.1	2.3	13.2	5.1	0.8	2.8	31.6	7.2	15.2	5.3	11.6	3.5	0.6	2.8	43.6	17.5
Cedar mill	2.8	3.5	41.5	0.9	0.2	2.1	14.1	35.1	1.9	4.1	34.7	1.2	0.2	2.1	11.3	44.5
Johnson	0.7	0.4	36.9	0.7	0.1	4.6	14.0	42.7	0.5	0.7	31.7	1.0	0.0	4.6	5.1	56.3
Quatama	27.6	0.3	45.1	0.2	0.2	2.7	18.4	5.6	21.3	1.3	47.1	0.4	0.3	2.7	17.4	9.5
RCHwy8	14.4	4.3	31.2	1.2	0.0	3.2	17.6	27.9	9.1	5.2	31.9	1.4	0.0	3.1	17.4	31.9
1000 m Rip																
BrnSalt	7.9	0.0	81.7	0.0	0.0	0.0	10.1	0.3	4.1	0.0	82.6	0.0	0.0	0.0	11.3	2.0
BrnWU	54.6	0.0	8.7	0.0	1.8	4.1	24.0	6.7	0.0	0.0	23.1	0.7	0.0	4.1	35.7	36.4
BrnBP	6.6	0.3	7.1	4.9	0.0	30.5	29.9	20.8	1.0	0.2	11.1	4.9	0.0	30.5	21.0	31.2
Bronson	0.0	2.4	0.0	6.3	0.9	20.2	44.8	25.4	0.0	1.1	3.2	9.1	0.4	20.2	24.1	41.9
Bvtn170	0.0	2.6	51.4	0.0	0.0	32.7	6.3	7.1	0.0	1.9	56.7	0.0	0.0	31.9	5.8	3.6
BvtnCP	1.0	0.0	21.0	0.0	0.0	0.0	41.3	36.7	0.0	0.0	21.5	0.0	0.0	0.0	43.6	34.9
DawAir	26.0	3.8	11.6	3.0	2.7	0.0	52.7	0.1	18.3	2.7	8.0	0.0	1.6	0.0	59.0	10.4
Dawson	17.6	0.0	37.2	0.4	0.0	6.2	3.6	34.9	11.3	0.0	30.2	0.3	0.0	6.2	13.5	38.5
Cedar mill	0.0	14.8	7.0	0.0	0.3	6.7	60.4	10.9	0.0	37.0	10.9	2.2	0.0	6.6	32.3	11.0
Johnson	0.0	0.0	20.5	3.2	0.0	12.7	4.0	59.5	0.0	0.6	18.3	4.0	0.0	12.7	1.8	62.6
Quatama	5.2	0.7	32.2	0.0	0.0	0.0	51.0	10.9	1.1	1.4	21.6	0.0	0.0	0.0	55.5	20.3
RCHwy8	6.5	0.1	41.6	1.4	0.0	3.0	39.7	7.8	1.9	0.0	30.9	0.0	0.0	2.7	50.2	14.4
500 m rip																
BrnSalt	11.7	0.0	75.9	0.0	0.0	0.0	12.0	0.5	5.9	0.0	76.1	0.0	0.0	0.0	15.1	2.9
BrnWU	42.8	0.0	9.2	0.0	1.2	7.9	36.2	2.6	0.0	0.0	25.4	1.4	0.0	7.8	28.0	37.4
BrnBP	7.0	0.5	7.9	7.9	0.0	1.5	43.3	31.9	1.7	0.3	15.5	8.0	0.0	24.4	21.8	28.4
Bronson	0.0	1.3	0.0	1.6	0.0	23.4	46.9	26.8	0.0	0.0	6.3	4.0	0.0	23.5	12.9	53.4
Bvtn170	0.0	0.0	70.0	0.0	0.0	26.7	0.0	3.4	0.0	0.0	73.4	0.0	0.0	25.9	0.0	0.7
BvtnCP	0.0	0.0	15.6	0.0	0.0	0.0	45.9	38.6	0.0	0.0	18.5	0.0	0.0	0.0	50.2	31.3
DawAir	23.0	0.0	24.4	5.4	6.7	0.0	40.3	0.1	11.2	0.3	19.9	0.0	3.0	0.0	55.0	10.6
Dawson	6.0	0.0	36.0	0.0	0.0	2.7	4.5	50.9	0.6	0.0	28.6	0.0	0.0	2.7	14.2	53.9
Cedar mill	0.0	9.6	5.4	0.0	0.0	5.6	63.5	15.9	0.0	42.1	2.0	3.2	0.0	5.6	27.0	20.1
Johnson	0.0	0.0	14.8	5.2	0.0	7.0	4.1	69.0	0.0	1.0	15.0	6.4	0.0	6.9	1.0	69.8
Quatama	0.0	0.0	35.5	0.0	0.0	0.0	28.9	35.6	0.0	0.0	40.1	0.0	0.0	0.0	26.0	33.9

Agri = Agriculture; Com = Commercial; Forst = Forest; Water = Water; Wet = Wetland; Res = Residential.

References

Anderson, J.R., Hardy, E.E., Roach, J.T., Witmer, R.E., 1976. A land use and land cover classification system for use with remote sensor data. In: U.S. Geological Survey Professional Paper, No. 964.

Band, L.E., Cadenasso, M., Grimmond, S., Grove, M., Pickett, S.T., 2005. Heterogeneity in urban ecosystems: pattern and process. In: Lovett, G.M., Jones, C.G., Turner, M.G., Weathers, K.C. (Eds.), Ecosystem Function in Heterogeneous Landscapes. Springer-Verlag, NY.

Bekele, A., McFarland, A., 2004. Regression-based flow adjustment procedures for trend analysis of water quality data. Transactions of the ASAE 47 (4), 1093–1104.

Booth, D.B., Jackson, C.R., 1997. Urbanization of aquatic systems: degradation thresholds, stormwater detection, and the limits of mitigation. Journal of the American Water Resources Association 33 (5), 1077–1090.

Chang, H., 2007. Streamflow characteristics in urbanizing basins in the Portland Metropolitan Area, OR, USA. Hydrological Processes 21 (2), 211–222.

Chang, H., Carlson, T., 2005. Water quality during winter storm events in Spring Creek, Pennsylvania, USA. Hydrobiologia 544, 321–332.

Choi, J.-Y., Engel, B.A., Muthukrshnan, S., Harbor, J., 2003. GIS based long term hydrologic impact evaluation for watershed urbanization. Journal of the American Water Resources Association 39 (3), 623–635.

City of Beaverton, Oregon, 2005. Beaverton Statistics. 〈http://www.beavertonoregon.gov/beaverton/statistics.html〉 (accessed 05.09.05).

Clean Water Services, 2005. Watershed Data Catalog. 〈http://www.cleanwaterservices.org/MapsAndData〉 (accessed 30.09.05).

Cleveland, W.S., 1979. Robust locally weighted regression and smoothing scatterplots. Journal of the American Statistical Association 74 (368), 829–836.

Cox, B.A., 2003. A review of dissolved oxygen modeling techniques for lowland rivers. The Science of the Total Environment 314–316, 303–334.

Creech, J., 2003. Urbanization and Its Relationship to Water Quality Within the Bronson Creek Watershed Master's Thesis, Portland State University, Portland.

Djodjic, F., Bergström, L., 2005. Phosphorus losses from arable fields in Sweden – effects of field-specific factors and long-term trends. Environmental Monitoring and Assessment 102 (1–3), 103–117.

Dunne, T., Leopold, L.B., 1978. Water in Environmental Planning. W.H. Freeman and Co., New York.

Earth System Research Institute (ESRI), 2005. ArcGIS 9.2. Redlands, CA.

Eaton, A.D., Clesceri, L.S., Greenberg, A.E. (Eds.), 1994. Standard methods for the examination of water and wastewater, 18 Ed. American Public Health Association, American Water Works Association, Water Environment Federation, Washington, D.C.

Esterby, S., 1996. Review of methods for the detection and estimation of trends with emphasis on water quality applications. Hydrological Processes 10 (2), 127–149.

García, L.V., 2003. Controlling the false discovery rate in ecological research. Trends in Ecology and Evolution 18 (11), 553–554.

Gilbert, R.O., 1987. Statistical Methods for Environmental Pollution Monitoring. Van Nostrand Reinhold, New York, NY.

Gove, N.E., Edwards, R.T., Conquest, L.L., 2001. Effects of scale on land use and water quality relationships: a longitudinal basin-wide perspective. Journal of the American Water Resources Association 37 (6), 1721–1734.

Hatt, B.E., Fletcher, T.D., Walsh, C.J., Taylor, S.L., 2004. The influence of urban density and drainage infrastructure on the concentrations and loads of pollutants in small streams. Environmental Management 34 (1), 112–124.

Heathwaite, A.L., Johnes, P.J., 1996. Contribution of nitrogen species and phosphorus fractions to stream water quality in agricultural catchments. Hydrological Processes 10 (7), 971–983.

Helsel, D.R., Hirsch, R.M., 1992. Statistical Methods in Water Resources. Elsevier, Amsterdam.

Hirsch, R.M., Slack, J.R., 1984. A nonparametric trend test for seasonal data with serial dependence. Water Resources Research 20 (6), 727–732.

Hirsch, R.M., Alexander, R.B., Smith, R.A., 1991. Selection of methods for the detection and estimation of trends in water quality. Water Resources Research 27 (5), 803–813.

Intelligent Decision Technologies, 1998. WQStat Plus v.1.56. Intelligent Decision Technologies, Ltd, Longmont.

Karr, J.R., Schlosser, I.J., 1978. Water resources and the land-water interface. Science 201 (4352), 229–234.

Lettenmaier, D.P., Hooper, E.R., Wagoner, C., Faris, K.B., 1991. Trends in stream quality in the continental United States, 1978–1987. Water Resources Research 27 (3), 327–339.

Massoud, M.A., El-Fadel, M., Scrimshaw, M.D., Lester, J.N., 2006. Factors influencing development of management strategies for the Abou Ali River in Lebanon II: Seasonal and annual variation. Science of the Total Environment 362 (1–3), 31–41.

McBride, M., Booth, D.B., 2005. Urban impacts on physical stream condition: Effects of spatial scale, connectivity, and longitudinal trends. Journal of the American Water Resources Association 41 (3), 565–579.

McBride, G.B., 2005. Using Statistical Methods for Water Quality Management: Issues, Problems and Solutions. Wiley, NY.

Morley, S.A., Karr, J.R., 2002. Assessing and restoring the health of urban streams in the Puget Sound basin. Conservation Biology 16 (6), 1498–1509.

Niyogi, D.K., Koren, M., Arbuckle, C.J., Townsend, C.R., 2007. Stream communities along a catchment land-use gradient: Subsidy-stress responses to pastoral development. Environmental Management 39 (2), 213–225.

Novotny, V., 2003. Water Quality: Diffuse Pollution and Watershed Management, Second Edition. Wiley, Hoboken.

Oregon Blue Book, 2005. City and County Populations. 〈http://bluebook.state.or.us/local/populations/populations.htm〉 (accessed 04.11.05).

Oregon Department of Environmental Quality, 2001. Tualatin Basin Total Maximum Daily Load (TMDL). Oregon Department of Environmental Quality, Portland.

Oregon Department of Environmental Quality, 2005. Water quality standards: Beneficial uses, policies, and criteria for Oregon. Oregon Administrative Rules, Chapter 340, Division 041.

Oregon Department of Environmental Quality, 2007. Water Quality Assessment. 〈http://www.deq.state.or.us/wq/assessment/assessment.htm〉, accessed 03.05.07.

Pan, Y., Herlihy, A., Kaufmann, P., Wigington, J., van Sickle, J., Moser, T., 2004. Linkages among land-use, water quality, physical habitat conditions and lotic diatom assemblages: a multi-spatial scale assessment. Hydrobiologia 515 (1–3), 59–73.

Passell, H.D., Dahm, C.N., Bedrick, E.J., 2004. Hydrological and geochemical trends and patterns in the upper Rio Grande, 1975–1999. Journal of the American Water Resources Association 40 (1), 111–127.

Räike, A., Pietiläinen, O.-P., Rekolainen, S., Kauppila, P., Pitkänen, H., Nieme, J., Raateland, A., Vuorenmaa, J., 2003. Trends of phosphorus, nitrogen and chlorophyll a concentrations in Finnish rivers and lakes in 1975–2000. The Science of the Total Environment 310 (1–3), 47–59.

Rounds, S.A., Doyle, M., 1997. Sediment Oxygen Demand in the Tualatin River basin, 1992–1996. US Geological Survey Water Resources Investigations Report 97-4103, Portland.

Scott, M.C., Helfman, G.S., McTammany, M.E., Benfield, E.F., Bolstad, P.V., 2002. Multiscale influences on physical and chemical stream conditions across blue ridge landscapes. Journal of the American Water Resources Association 38 (5), 1379–1392.

Sliva, L., Williams, D.D., 2001. Buffer zone versus whole catchment approaches to studying land use impact on river water quality. Water Research 35 (14), 3462–3472.

Snyder, C.D., Young, J.A., Villella, R., Lemarie, D.P., 2003. Influences of upland and riparian land use patterns on stream biotic integrity. Landscape Ecology 18 (7), 647–664.

Stansfield, B., 2001. Effects of sampling frequency and laboratory detection limits on the determination of time series water quality trends. New Zealand Journal of Marine and Freshwater Research 35 (5), 1071–1075.

Stewart, J.S., Wang, L., Lyons, J., Horwatich, J.A., Bannerman, R., 2001. Influences of watershed, riparian-corridor, and reach-scale characteristics on aquatic biota in agricultural watersheds. Journal of the American Water Resources Association 37 (6), 1475–1487.

Tobin, G.A., Brinkmann, R., 2002. The effectiveness of street sweepers in removing pollutants from road surfaces in Florida. Journal of Environmental Science and Health Part A – Toxic/Hazardous substances & Environmental Engineering 37 (9), 1687–1700.

Tong, S.T.Y., Chen, W.L., 2002. Modeling the relationship between land use and surface water quality. Journal of Environmental Management 66 (4), 377–393.

US Environmental Protection Agency (USEPA), 2007. Clean Water Act. http://epa.gov/region5/water/cwa.htm, (accessed, 03.05.07).

US Geological Survey (USGS), 〈www.nwis.waterdata.usgs.gov〉 (accessed, 05.09.05).

US Census Bureau, 2005. United States Census 2000. 〈http://www.census.gov/main/www/cen2000.html〉, (assessed, 05.09.05).

Wang, X., 2001. Integrating water quality management and land-use planning in a watershed context. Journal of Environmental Management 61 (1), 25–36.

Wilson, D.C., Burns, S.F., Jarrell, W., Lester, A., Larson, E., 1999. Natural ground-water discharge of orthophosphate in the Tualatin Basin, northwest Oregon. Environmental and Engineering Geosciences 5 (2), 189–197.

Yu, Y.-S., Zou, S., Whittemore, D., 1993. Non-parametric trend analysis of water quality data of rivers in Kansas. Journal of Hydrology 150 (1), 61–80.

Zipper, C.E., Holtzman, G.I., Darken, P.F., Gildea, J.J., Stewart, R.E., 2002. Virginia USA water quality, 1978 to 1995: Regional interpretation. Journal of the American Water Resources Association 38 (3), 789–802.

ELSEVIER

Journal of Environmental Management 87 (2008) 582–590

Journal of
Environmental
Management

www.elsevier.com/locate/jenvman

Characteristics and significance of liquid effluent from woodchip corrals in Scotland

A.T. McDonald[a,*], A.D.J. McDonald[b], D. Kay[b], J. Watkins[c]

[a] *Earth and Biosphere Institute, University of Leeds, Leeds LS2 9JT, UK*
[b] *River Basin Dynamics and Hydrology Research Group, IGES, University of Wales, Aberystwyth SY23 3DB, UK*
[c] *CREH Analytical Ltd., 50 Back Lane, Horsforth, Leeds, West Yorkshire LS18 4RS, UK*

Received 29 January 2007; received in revised form 30 March 2007; accepted 10 April 2007

Abstract

Woodchip corrals are increasingly used as cost effective means of over-wintering livestock in temperate regions but there is little information on their potential environmental impact. Four woodchip corrals of varying characteristics were instrumented to capture and quantify the flows reaching the base, where pollutant fluxes may move either vertically to groundwater, or laterally to a water course. Samples for chemical analysis were collected daily by auto-sampler. Samples for bacterial analysis were aseptically hand-sampled. Sampling frequency was increased during high flow events and sampling was conducted over a 12-month period. Microbiological samples were analysed for total coliform (TC), presumptive *Escherichia coli* (EC) and intestinal enterococci (IE). Leachate was also analysed for total phosphorus, phosphate, total nitrogen, ammonium, total oxidised nitrogen, nitrite and nitrate. Each corral had a recording rain gauge sited within 10 m of the corral surface. Mean total nitrogen concentration in leachate was 339.5 mg l^{-1}, of which ammoniacal-N comprised approximately 57%. Mean total phosphorus concentration was 94.7 mg l^{-1}. Geometric mean concentrations of TC, EC and IE were 95,461, 94,983 and 55,552 cfu 100 ml^{-1}, respectively. Significant flows of leachate occurred at the base of the corrals on most days during the 1-year sampling period and flow rate increased with stocking density. Strong positive linear relationships were found between the concentrations of the nutrient parameters and discharge. Strong positive curvilinear relations were found between faecal indicator concentrations and discharge. Different relationships were observed in the stocked and unstocked corrals. The resulting fluxes are sufficient to give concern and to indicate that corral development is worthy of regulatory attention.

Keywords: Woodchip corrals; Effluent; Nitrogen; Phosphorus; Coliform; *E. coli*; Enterococci; Seasonality; Concentration; Fluxes

1. Introduction

Woodchip corrals provide a means of over-wintering cattle and other livestock outdoors. Woodchip corrals, having low capital and operational costs, are regarded as a cost effective option in Scotland and England and may offer animal welfare and environmental benefits. The Scottish Agricultural College suggested in 2000 that the capital cost was £70–£90 per cow with an annual upkeep of £5–£10, significantly lower than the cost of housing and mucking out. The use of woodchip corrals started in New Zealand and appears to have spread rapidly to Scotland and Ireland and more slowly to England. The welfare and environmental benefits are said to include a drier softer surface for the beasts, fewer of the respiratory problems associated with enclosed housing, reduced soil poaching, with a consequent improvement in the availability of spring grazing, and potential within corral 'treatment' of organic wastes. There is no robust scientific evidence to support or refute these suggestions, particularly those relating to within corral treatment of wastes. Guidance to the farming community is predominantly found on web sites and in trade journals (Edwards et al., 2003). These have generally advocated top-soil removal prior to corral construction. However, there is no accepted 'best practice' and evidence-based scientific advice on appropriate

---

[*] Corresponding author. Tel.: +44 113 343 3344.

E-mail addresses: a.t.mcdonald@leeds.ac.uk (A.T. McDonald), dave@crehkay.demon.co.uk (D. Kay).

0301-4797/$ - see front matter
doi:10.1016/j.jenvman.2007.04.029

construction to minimise environmental impact is lacking. Although there is no definitive information on the number of corrals in the UK, the number in Scotland is commonly thought to be at least 500–600. The Scottish Agricultural Pollution Group (SEPA, 2003) quoted the number as over 400 in 2002, a time at which numbers were likely to be expanding rapidly as corrals were being extensively publicised and the numbers in England and Wales are thought to be similar. Thus, with perhaps 1000 such sites in the UK, the information deficit relating to their efficacy and environmental performance requires to be addressed.

In the United Kingdom, corrals are usually stocked from November to April, although this varies with weather conditions and business priorities and some corrals are used as a holding site for livestock over the summer period. Constructed corrals exhibit great diversity in terms of area, stocking density, boundary construction, woodchip depth, size and nature of woodchips, drainage, maintenance and life cycle. However, there is no systematic data acquisition to facilitate quantification of this diversity in existing systems.

2. Materials and methods

This study examined four sites in livestock rearing areas of Scotland. The sites were paired for logistic purposes but are not a paired design in the sense of a longitudinal study involving an experimental and a matched control site. Rather, the sites were selected with the involvement of stakeholders to be broadly representative of corrals in Scotland. One pair was sited in north-east Scotland (NE1 and NE2) with a second pair in south west Scotland (SW1 and SW2) which provided a comparison of a higher rainfall maritime western margin climate with a dryer eastern climatic zone.

In three of the four sites, the following site instrumentation protocol was employed as illustrated in Fig. 1. An area of woodchip was cleared to the base of the corral. A timber frame, adapted to the micro-topography of each site, was used to define the sample area and a multi-layer impermeable geomembrane was attached to the frame for leachate collection. This was sited in a representative area of the corral and recovered with the original woodchip to the same depth as the rest of the corral surface. The leachate was piped to locked weatherproof instrumentation station where flow was measured by tipping buckets fitted with transducers linked to a data logger. Flow entered a collector where discrete samples were taken by ISCO autosamplers sampling at preset intervals. Temperature was continuously recorded. Rainfall was recorded by tipping bucket rain gauge sited within 10 m of each corral.

This sample-capture protocol was employed at sites SW2, NE1 and NE2. The approach has the advantage of precise measurement of the capture area for the sampling but also disturbs any established stratification in the matrix and, hence potentially disrupts biologically active 'treatment' of leachate moving down through the woodchips. This possibility is not encountered at site SW2 which was a new corral constructed with installed instrumentation. At site SW1, a 'non-disturbance' protocol was adopted with the leachate captured at the edge of the corral where it flowed downslope along the regolith beneath the cleared soil to a convenient collection location. Clearly, at this site, the 'contributing area 'is ill-defined and, thus, discharge per unit area cannot be assessed with the same confidence. However leachate quality which is characteristic of an undisturbed matrix is of interest to the regulatory community and differences between the flows from the disturbed and undisturbed sites will be valuable in the development of our understanding of corral maturation processes.

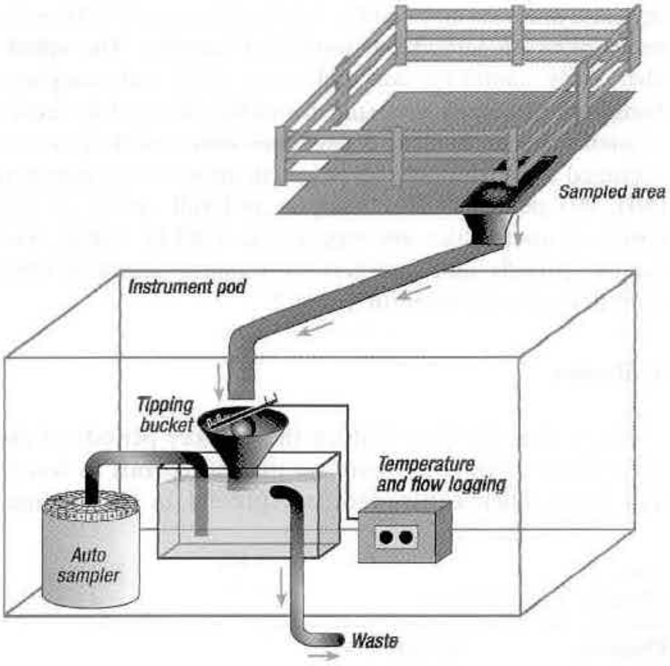


Fig. 1. Schematic of the sampling system employed.

The following data were recorded: (i) rainfall and outflow volume, (ii) nutrient status (by automated sampling every 24 h), (iii) faecal indicator organism concentrations (by hand sampling every 2 weeks) and, finally, (iv) 'event' sampling to capture high rainfall conditions at hourly intervals for chemistry and faecal indicators.

Samples for bacteriological analysis were collected aseptically using 150 ml sterile plastic containers (Media Disposables^{TM}). Samples were stored in cool boxes containing ice packs, and subsequently transported from field sites to the CREH laboratory. On receipt, samples were refrigerated at 2–8 °C in the dark and analysed within 24 h of field collection. Samples were analysed for faecal indicator organisms (FIOs) following standard UK methods (SCA, 2000, 2002). One sample from each run was analysed in duplicate to provide a quality control for each enumeration method.

Chemical analyses were conducted as detailed in Table 1. Full quality control procedures were employed including replicate analyses and blanks. All analyses, except pH, were undertaken on settled but unfiltered samples. The scheduled daily sampling was achieved using autosamplers. Some sample decay and transformation inevitably occurs in such systems. Sample decay rates were determined and averaged approximately 0.5% loss in ammoniacal nitrogen (NH_4–N) per day. This analysis and full details of the corral characteristics are reported in CREH (2005). The sample periods and numbers of samples acquired from each field site are given in Table 2.

3. Results

The results are presented for the two key periods of the year, those when cattle were on the test corrals in winter and those when cattle were not present in the summer period. While these periods are broadly the same across the industry and the region, the specific dates of usage of the corrals are controlled by individual farmers. Here, the four corrals are managed by three different farmers. Table 2 indicates the dates of corral occupancy which was broadly from November to April. Table 3 gives summary information for nutrient concentrations during the 'cattle-on' and 'cattle-off' periods. Table 4 gives summary information for FIO during the 'cattle-on' and 'cattle-off' periods.

Table 1
Chemical analysis methods

Chemical	Method
pH	Measured directly using a Hanna HI 9024 pH meter
Total nitrogen	Thermal combustion with chemiluminescence detection
Dissolved organic carbon	Thermal combustion with IR detection
Total phosphorus	Digested using Rowland and Haygarth's acid persulphate method as described in Rowland and Haygarth (1997). Analysed colorimetrically using a Skalar SAN^{++} Analyser with the modified Murphy and Riley method
Nutrient analysis	(Nitrate, nitrite, phosphate, ammonium): analysed using a Skalar SAN^{++} Analyser. See below for more details
Nitrate plus nitrite	Measured colorimetrically as nitrite (NAD-coupled diazotized sulphanilamide) following hydrazine reduction
Nitrite	Measured colorimetrically as NAD-coupled diazotized sulphanilamide, allowing the determination of nitrate by subtraction
Ammonium	Measured colorimetrically using the modified Berthelot (indophenol) reaction as described in Searle (1984)
Phosphate	Measured colorimetrically with the modified Murphy and Riley (1962) as Molybdate Reactive Phosphorus (MRP)

Since there are two management periods in corral operation, the period when livestock are present on the corral and the period when they are absent which coincide with winter and summer, respectively, the periods are characterised by differences in water balance as well as differences in input loading. Flow from sites SW1, SW2 and NE1 are significantly reduced in the summer cattle off periods to 25–75% of the winter flow values per unit of corral surface. The much more lightly stocked NE2 had significantly lower winter flows which reduced by a more modest 20% in the summer cattle-off period. However the good condition of the surface of this latter corral may have encouraged the retention of cattle-on the corral for a further month making direct comparisons more complex.

The 'cattle-off' period shows generally lower values for all determinants. Fig. 2 shows the dissolved organic carbon concentrations observed in SW2 outflow leachate during the entire sampling period. This is typical of the individual data runs for many of the chemical and microbiological parameters summarised in Tables 3–6. The higher, 'cattle-on', values, typically two to three times the 'cattle-off' values, suggest that the winter period will be the time of highest risk to water quality, particularly in relation to nitrogen flux. For example, total nitrogen concentration at site SW1 averaged $584.7\,\mathrm{mg\,l^{-1}}$ during 'cattle-on' conditions and $263.3\,\mathrm{mg\,l^{-1}}$ during 'cattle-off' conditions. This is an established corral sampled with a no disturbance approach. Disturbance in sampling or immaturity of the corral cannot be an explanation of the high values found. Ammoniacal nitrogen in SW2 (a new build corral sampled in the first season of operation with no sampling disturbance as the monitoring system was built in with the corral) averaged $234.0\,\mathrm{mg\,l^{-1}}$ and $94.9\,\mathrm{mg\,l^{-1}}$ in 'cattle-on' and 'cattle-off' conditions, respectively.

Table 2
Sampling site characteristics, dates, periods and numbers

	SW1	SW2	NE1	NE2
First FIO sample	2 Nov 2003	2 Nov 2003	18 Dec 2003	18 Dec 2003
First nutrient sample	21 Nov 2003	04 Dec 2003	01 Dec 2003	01 Dec 2003
Sampling end date	29 Sep 2004	29 Sep 2004	29 Sep 2004	29 Sep 2004
Nutrient sample	384	401	326	305
FIO sample	170	158	101	111
Cattle-on	Nov 2004	Nov 2004	Nov 2004	Dec 2004
Cattle-off	April 2004	April 2004	April 2004	May 2004
Stocking density head per $1000\,\mathrm{m^2}$	115	115	65	43
Woodchip depth (m)	0.6	0.6	0.9	0.3

Table 3
Summary of chemical parameters (concentrations and fluxes) and flows from four woodchip corrals in 2004–2006; sample numbers and sampling periods are the same as given in Table 2

		SW1	SW2	NE1	NE2
Corral and sample attributes	Total surface area of corral (m^2)	1282	1300	1846	370
	Sampled area of corral (m^2)	200	23	11.61	13.53
Cattle-on period flow characteristics	Averaged pipe outflow (period) (l h^{-1})	39.92	4.71	2.48	0.66
	Averaged flow per m^2 of corral (l h^{-1} m^{-2})	0.20	0.20	0.21	0.05
	Total corral flow (l h^{-1})	255.90	265.47	394.41	18.18
Cattle-on period nutrient concentrations	Total nitrogen (mg l^{-1})	584.7	532.5	213.6	589.1
	Ammonium – N (mg l^{-1})	372.0	234.0	76.1	399.1
	TON – N (mg l^{-1})	2.8	3.5	3.4	1.7
	Nitrite – N (mg l^{-1})	2.0	2.1	0.4	0.8
	Total phosphorus – P (mg l^{-1})	246.9	89.1	37.86	76.3
	Phosphate – P (mg l^{-1})	58.2	52.5	27.1	36.0
	DOC (mg l^{-1})	4312.3	3815.0	1578.4	2556.8
	pH	7.7	7.7	7.8	7.9
Cattle-on period nutrient load per unit area	Total nitrogen (mg h^{-1} m^{-2})	114.7	98.7	25.7	10.6
	Ammonium – N (mg h^{-1} m^{-2})	75.6	35.3	9.3	4.3
	TON – N (mg h^{-1} m^{-2})	0.47	1.87	0.39	0.06
	Nitrite – N (mg h^{-1} m^{-2})	0.45	0.48	0.05	0.03
	Total phosphorus – P (mg h^{-1} m^{-2})	20.3	18.7	3.7	2.3
	Phosphate – P (mg h^{-1} m^{-2})	12.8	13.2	2.8	1.5
	DOC (mg h^{-1} m^{-2})	941.6	837.5	179.2	94.2
Cattle-off period flow characteristics	Averaged pipe outflow (period) (l h^{-1})	153.15	3.10	0.57	0.56
	Averaged flow per m^2 of corral (l h^{-1} m^{-2})	0.15	0.13	0.05	0.04
	Total corral flow (l h^{-1})	196.34	175.03	91.35	15.19
Cattle-off period nutrient concentrations	Total nitrogen (mg l^{-1})	263.3	177.5	38.42	99.2
	Ammonium – N (mg l^{-1})	194.4	94.9	4.64	22.3
	TON – N (mg l^{-1})	10.2	5.5	4.0	2.1
	Nitrite – N (mg l^{-1})	4.4	2.1	0.3	0.7
	Total phosphorus – P (mg l^{-1})	62.4	85.3	90.2	80.9
	Phosphate – P (mg l^{-1})	52.7	49.2	69.0	65.4
	DOC (mg l^{-1})	1987.0	1759.9	875.8	1305.5
	pH	8.0	8.0	8.3	8.0
Cattle-off period nutrient load per unit area	Total nitrogen (mg h^{-1} m^{-2})	19.78	11.75	2.89	1.76
	Ammonium – N (mg h^{-1} m^{-2})	13.14	4.33	0.32	0.25
	TON – N (mg h^{-1} m^{-2})	1.66	1.25	0.25	0.09
	Nitrite – N (mg h^{-1} m^{-2})	0.14	0.10	0.02	0.03
	Total phosphorus – P (mg h^{-1} m^{-2})	10.97	5.50	3.46	2.85
	Phosphate – P (mg h^{-1} m^{-2})	9.76	4.69	2.88	2.39
	DOC (mg h^{-1} m^{-2})	165.4	168.8	46.4	33.0

However, the 'cattle-off' period is also typified by substantial leachate concentrations of total nitrogen, ammoniacal nitrogen, total phosphorus and phosphate despite the changed water balance and input regimes. For example, NE1 in the 'cattle-off' period had mean values of 38.4 mg l^{-1}, 4.64 mg l^{-1}, 90.2 mg l^{-1} and 69.0 mg l^{-1}, respectively for these determinants. Further, the 'cattle-off' period does not appear to show a steady decline in any of the cases but, rather, a discrete change to a new state. This suggests that options such as a longer cattle-off period are not likely to reduce these flows to an acceptable level. Fig. 2 shows throughout the 'cattle-off' period, no change in the leachate concentration which ranged between 1000 and 2000 mg DOC l^{-1}.

The relationships between concentrations and flow reveal a slight dilution effect that occurs in both the 'cattle-on' and 'cattle-off' periods. This is illustrated in Fig. 3 for total nitrogen. At higher flows, generally lower concentrations of total nitrogen are observed. At lower flows, the concentrations experienced are highly variable but include a significant number of higher values. The dilution effect is more pronounced in the 'cattle-off' period. During the 'cattle-on' period the concentration of total nitrogen at higher flows is an order of magnitude greater than those observed at high flow in the 'cattle-off' period. However, at all times and under all flow conditions, high concentrations reach the corral base and are evident in the measured leachate.

Since flow driven responses are frequently important in water quality studies, Tables 5 provide geometric means for FIO samples split into 'cattle-on' and 'cattle-off' periods under both low and high flow conditions. Concentration

Table 4
Summary FIO expressed as log $_{10}$ geometric mean concentrations

		SW1	SW2	NE1	NE2
Cattle-on period faecal indicator organism concentrations	Total coliforms, TC (cfu 100 ml^{-1})	5.4585	5.6060	4.8537	6.5370
	Escherichia coli, EC (cfu 100 ml^{-1})	5.2115	5.4496	4.4992	6.4538
	Intestinal enterococci, IE (cfu 100 ml^{-1})	5.2529	5.5562	5.3937	6.9821
Cattle-on period faecal indicator organism load per unit area	Total coliforms, TC (cfu h^{-1} m^{-2})	5.8138	5.6809	5.0416	6.1342
	Escherichia coli, EC (cfu h^{-1} m^{-2})	5.5834	5.5831	4.7933	6.0258
	Intestinal enterococci, IE (cfu h^{-1} m^{-2})	5.3370	5.6894	5.5372	6.5058
Cattle-off period faecal indicator organism concentrations	Total coliforms, TC (cfu 100 ml^{-1})	4.2752	4.1072	4.0783	4.5774
	Escherichia coli, EC (cfu 100 ml^{-1})	3.8192	3.9169	3.9774	4.3917
	Intestinal enterococci, IE (cfu 100 ml^{-1})	3.5160	3.9743	3.2686	4.9466
Cattle-off period faecal indicator organism load per unit area	Total coliforms, TC (cfu h^{-1} m^{-2})	4.1155	4.1113	4.2454	4.0896
	Escherichia coli, EC (cfu h^{-1} m^{-2})	3.7280	4.0478	3.5481	3.9039
	Intestinal enterococci, IE (cfu h^{-1} m^{-2})	3.4936	4.0329	3.4954	4.4588

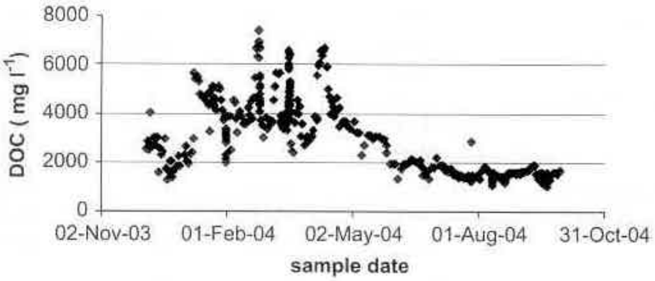


Fig. 2. Site SW2 all records – DOC chemograph.

data alone, however, will not characterise the environmental loading of woodchip corral leachates escaping to the wider environment and thus Table 6 reports fluxes of FIO, again for flow conditions and management period. Geometric means (i.e. the antilog of the arithmetic mean of the $\log_{10}$ values derived from each FIO enumeration) are the accepted and most statistically appropriate central measure of central tendency for the right skewed distributions characteristic of FIO data acquired from environmental aqueous matrices (Kay et al., 2005, 2007). $\log_{10}$ transformation produces a probability density function which more closely approximates to normality than the raw data, i.e. generally, producing data more appropriate for parametric statistical tests such as analysis of variance. This is true of stream, marine and sewage effluent samples (Kay et al., in press; Stapleton et al., in this issue) which have been initially transformed to $\log_{10}$ values.

4. Discussion

Drenagh Sawmills has been one of the strongest industry advocates of woodchip corrals. They provide (http://www.drenagh.co.uk/indexc.htm) an analysis of waters from a drain beneath a woodchip corral. No detail is given on the nature of the corral, stocking conditions, the time of sampling or the methods of analysis. Nevertheless it is instructive to compare the analysis reported here with the figures from Drenagh. They report nitrite, nitrate, ammonium and phosphate as 3.4, 2, 250 and 130 mg l^{-1} respectively. These values accord broadly with the values reported in this study. Drenagh reports a COD of 4400 mg l^{-1} and estimates BOD to be 50–75% of the COD value. The current research has used DOC, which equates broadly with ultimate BOD, and average DOC ranged from 1578 to 4312 mg l^{-1} across the four sites. Again the outcomes are broadly comparable. Assuming that a company supporting woodchip corrals would sample from a well-run corral, the results support the view that the sampled sites in this paper are indeed sites typical of well-run operations and so the poor leachates cannot be dismissed as the product of a poorly run operation. US Soil Conservation Service reports the chemical characteristics of the waters from feedlot ponds as having 1.67% total nitrogen the majority being ammonium, which is a similarly high but not readily comparable figure.

Environmental loading is better characterised by flux than by concentration alone. Fig. 4 illustrates the relationship between flux and rate of flow again for site SW2. The dilution effects observed in Fig. 3 do not reduce the flux at high flow which is increased due to the elevated discharge. This positive relationship is seen in both the 'cattle-on' and 'cattle-off' seasons. Thus, in summer (i.e. 'cattle-off') high flows following rainfall events, high fluxes were observed although these were lower than those observed under similar flows in the 'cattle-on' condition. Since flux is the product of volume and concentration values it is not appropriate to display r^2 for these relationships.

Similar relationships were observed for all major pollutants quantified in the study, both chemical and microbiological parameters. To illustrate this, Figs. 5 and 6 present data for total coliforms for both management periods and the flow control of flux is evident suggesting a 'transport limited' rather than 'supply limited' export process under the Scottish climatic regime. Fig. 7 summarises data for the microbiological parameters and

Table 5
FIO concentrations expressed as $\log_{10}$ geometric means for high and low flow conditions during the 'cattle-on' and 'cattle-off' periods

Period	Flow condition	'Cattle-on'	SW1	SW2	NE1	NE2
Cattle-on	High flow	Total coliforms (TC)	5.4138	5.6319	5.4395	6.6102
		Escherichia coli (EC)	5.1729	5.5477	5.2414	6.5107
		Intestinal enterococci (IE)	4.9181	5.4347	5.6723	7.0096
		Samples	45	47	32	13
	Low flow	Total coliforms (TC)	5.5720	5.4003	4.4024	6.5511
		Escherichia coli (EC)	5.3493	5.2711	4.2177	6.3821
		Intestinal enterococci (IE)	5.0011	5.3394	5.0798	6.9031
		Samples	33	29	29	28
Cattle-off	High flow	Total coliforms (TC)	5.4469	4.6626	5.4060	5.3245
		Escherichia coli (EC)	3.8814	4.2996	5.1926	5.1474
		Intestinal enterococci (IE)	4.3813	4.2654	4.9837	5.0074
		Samples	18	37	7	21
	Low flow	Total coliforms (TC)	3.5023	3.8973	3.2745	3.7273
		Escherichia coli (EC)	3.5482	3.7698	2.7760	3.3419
		Intestinal enterococci (IE)	3.0745	3.9597	2.4328	4.9258
		Samples	31	35	15	10

Sample numbers are given for each site under each of the four tabulated conditions.

Table 6
FIO fluxes expressed as geometric means ($\log_{10}$ cfu h^{-1} m^{-2}) and average outflow (l h^{-1} m^{-2}) expressed per square meter of corral area for dry and wet periods during the 'cattle-on' and 'cattle-off' periods

Period	Flow	Measured variable	SW1	SW2	NE1	NE2
'Cattle-on'	Low	Total coliforms	5.5538	6.1258	4.5798	5.4301
	Low	*Escherichia coli*	5.3311	6.0311	4.3952	5.2611
	Low	Intestinal enterococci	4.9829	6.1255	5.2573	5.7821
	Low	Flow per m^2	0.1204	0.6354	0.2467	0.0104
	High	Total coliforms	5.6952	7.0206	6.5880	6.8285
	High	*Escherichia coli*	5.4543	6.9364	6.3899	6.7290
	High	Intestinal enterococci	5.1994	6.8234	6.8209	7.2279
	High	Flow per m^2	0.2475	2.7112	1.9871	0.4412
'Cattle-off'	Low	Total coliforms	3.0164	4.1304	2.9919	2.2687
	Low	*Escherichia coli*	3.0623	4.1648	1.9876	1.8833
	Low	Intestinal enterococci	2.5886	4.2693	1.6499	3.4672
	Low	Flow per m^2	0.0355	0.2387	0.0169	0.0059
	High	Total coliforms	6.1336	6.5172	6.8717	6.3304
	High	*Escherichia coli*	4.5681	6.1663	6.6583	6.1533
	High	Intestinal enterococci	5.0680	6.1204	6.4494	6.0133
	High	Flow per m^2	0.6896	8.8147	3.3586	2.3550

reinforces the generic nature of the outcomes. Here, the data are expressed as flux per unit area of corral.

It has been asserted that aging and maturation of corrals may be required before the corral can function as an effective water treatment medium and reduce the chemical and microbiological load in the effluent. Site SW1, a mature established corral, was instrumented using a non-disturbance approach to preserve stratification and 'maturity'. Neither chemical nor microbiological parameters from SW1, expressed as concentrations, fluxes or flux per unit area, showed any improvement. Indeed, with the modest exception of phosphorus during the cattle-off period, SW1 appears to function less effectively than the disturbed corrals. This adds some weight to the proposals that, as part of the management regime, corrals need to be aerated by turning the surface during the summer period.

Corrals are alleged to 'self-seal' the implication being that there is negligible flow to the subsoil and to ground waters. The runoff volumes reported in this study suggest that complete self sealing is unlikely to occur. The simple mass balance of liquid under self-sealing conditions would have resulted in obvious, prolonged and substantial flow from the corrals across the ground surface which was not observed in any of the sites. The study has focussed only on the liquid losses to the environment and has made no measure of the gaseous pathway that will be a source of further loss from the corral. In addition if corrals trap a significant volume of water during heavy rainfall periods

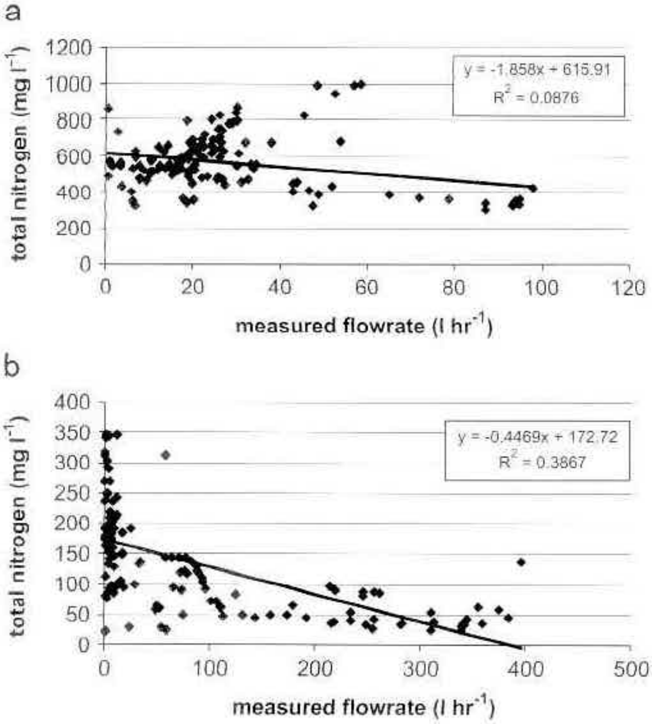


Fig. 3. Site SW2 'cattle-on' period (a) and 'cattle-off' period (b) total nitrogen concentration vs. measured flow rate.

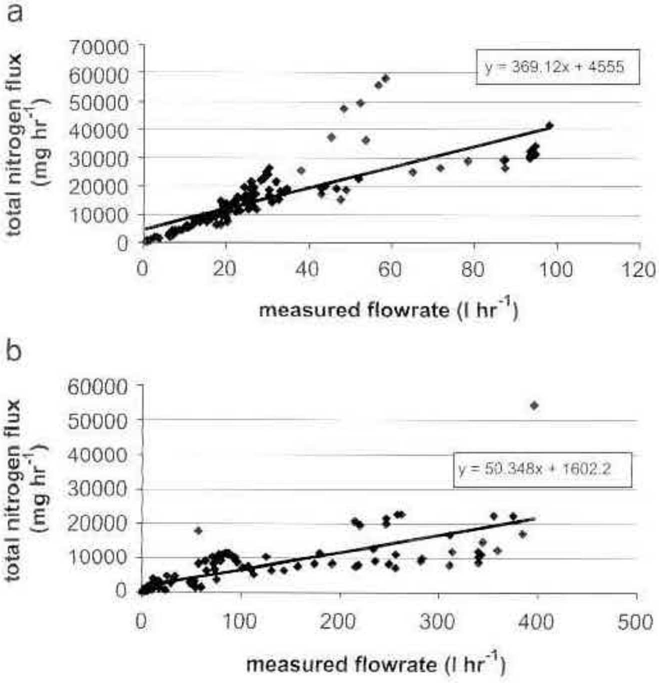


Fig. 4. Site SW2 'cattle-on' period (a) and 'cattle-off' period (b) total nitrogen flux vs. measured flowrate.

they may also be subject to liquefaction and mass movements with consequent loss of solid material. Such events were observed on two corrals but were not quantified. Therefore the significant impacts represented by the chemical and bacterial fluxes reported here are likely to be a major but not the total measure of impacts.

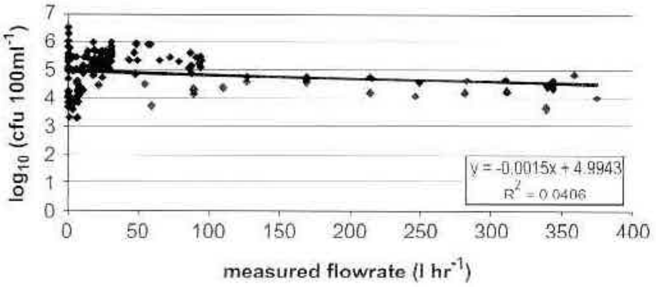


Fig. 5. Site SW2 total coliform (TC) concentration against measured flowrate.

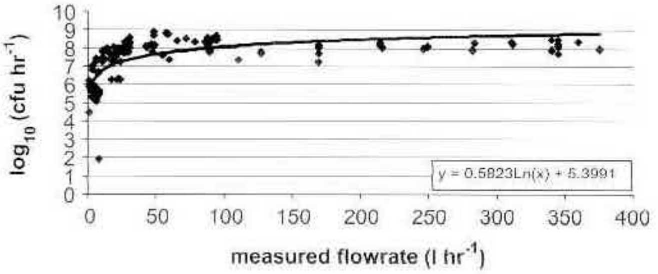


Fig. 6. Site SW2 all data – total coliform (TC) fluxes against flowrate.

Proponents of corrals suggest that the feed site should be outside the corrals and that since the cattle will defecate more while feeding, a sloping ramp that can be scraped clean to a treatment pit should be installed. This arrangement exists at site NE1 and may explain the rather better quality of the leachate in comparison to the more lightly stocked NE2. However, since NE2 also has a thinner woodchip cover this factor might also partially explain the differences. The sloping ramp on NE1 may, however, also be responsible for the mass movement of corral contents observed at that site.

5. Conclusions

The results clearly indicate that the four corrals monitored in both the stocked and unstocked condition do not 'treat' the nutrient or faecal indicator 'inputs' to produce outflows which are environmentally benign. Indeed, empirical data presented here suggests that there is high strength leachate leaving the base of all the corrals throughout the year.

The flux of pollutants is controlled by flow rate and adjusting the water balance may be the key lever in corral management. For corrals in the environments studied, there are three actions that could be considered, namely: (i) constructing an impermeable base membrane with liquid capture, containment and treatment, (ii) roofing of corrals, and/or (iii) limitations on corral stocking density and increased corral depth. Of these three options, only the first provides certainty that effluents will not escape the corral to the wider aquatic environment. However the cost of the

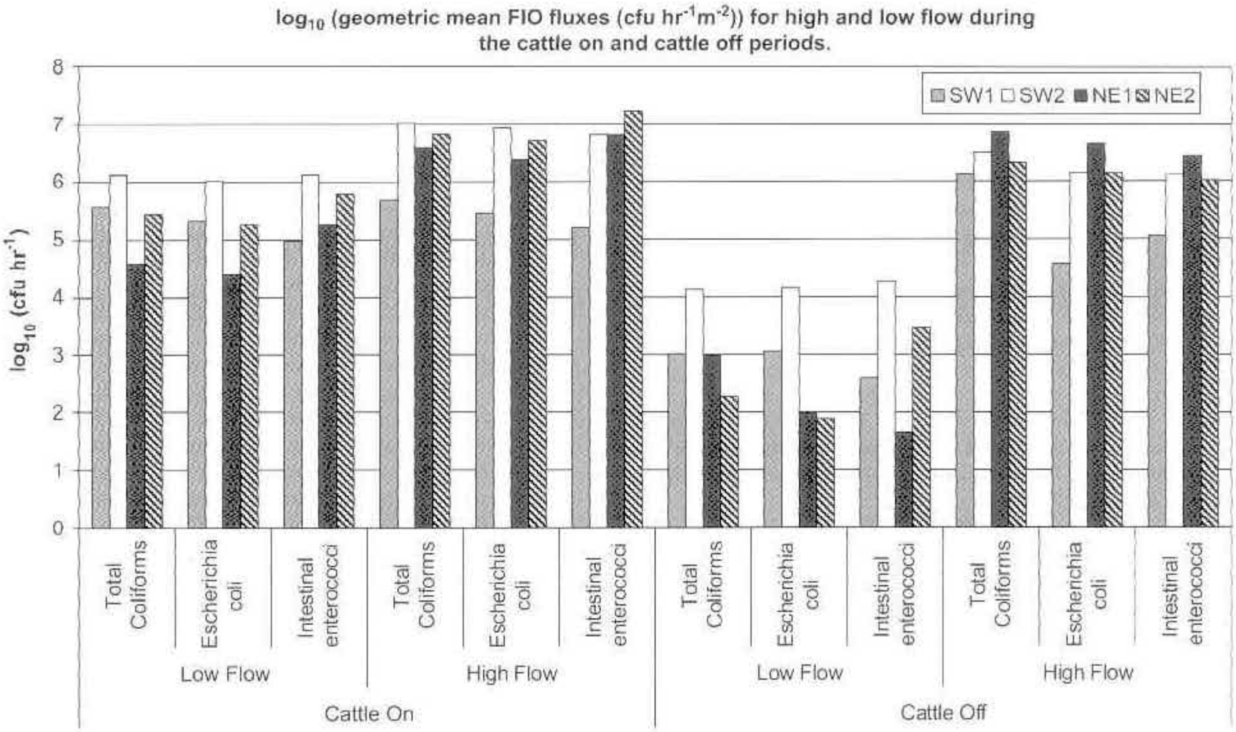


Fig. 7. Faecal indicator organism fluxes (as $\log_{10}$ cfu h^{-1}) for all sites, flow conditions and management periods.

required effluent treatment will partially reduce the economic benefit sought through corrals which require no cleaning or treatment in comparison to traditional over-wintering in sheds.

Some corrals on deep sandy soils, of the type reported in Vinten et al. (2006), may be able to utilise the natural filtration and purification capacity of the soil ecosystem to reduce the bacterial flux but such sites will remain a relatively small proportion of the land area under pastoral agriculture in northwest Europe. Further, such soils appear to exercise little control on the passage of nitrate to groundwater. The recent milder winters in the UK may be extending the period during which cattle may be grazed outdoors. This will reduce the period during which cattle are held on woodchip corrals and, with lower stocking, may delay the onset of saturation thus reducing flows to the environment. The longer unstocked period may allow corral recovery if allied to opening the corral surface during the summer period. The problem here for the regulator is that the outcome is uncertain depending on weather conditions which control both the duration of corral residence and the likelihood of saturation. The key question is an economic one, however. Will corrals as an alternative to sheds remain viable if liquor has to be captured and treated or corral depth and stocking increased and decreased, respectively?

The future of corrals lies in holistic integrated management and careful site selection. ADAS, in a report to the Environment Agency, identified several strategies that must be implemented if corrals are to be less polluting and remain effective agricultural management options. Corrals need to be carefully designed from the outset, should be on sites that will dry out rapidly and should not be sited in areas of high water table. Compacted dung and woodchip near the feed areas needs to be removed each year and composted in accordance with EA protocols (EA, 2007). A high strength leachate should be expected at most sites. This could be captured but may be treated through integrated reed bed and managed wetland systems. The corrals need to be treated in the context of a further potential source of diffuse agricultural pollution (Scottish Executive, 2005) and mitigated as other potential sources (Kay et al., 2005).

Acknowledgements

This investigation could not have been completed without the support, guidance and cooperation of a number of individuals and organisations. We wish formally to acknowledge the comments, observations, analysis and guidance of the Scottish Executive Environmental and Rural Affairs Department steering group chaired by Martin Johnston. Several organisations were particularly helpful in our search for representative sites. We wish to thank in particular staff at MLURI, SEPA and SNH. Dr Tony Edwards gave invaluable scientific comment. We wish to record, with our gratitude, the support, cooperation and practical interventions of the farmers involved and of their staff. Errors of fact and interpretation remain, however, with the authors.

References

CREH, 2005. Flow and Leachate Characteristics of Four Woodchip Corrals in Scotland. Report to the Scottish Executive Environment and Rural Affairs Department. Available from: 〈http://www.scotland.gov.uk/Publications/2005/09/26120338/03393〉.

Edwards, A.C., Campbell, C.D., Paterson, E., 2003. Review of the Scientific Rationale Underpinning the Establishment of Woodchip Corrals. Macaulay Institute, Scotland, UK.

Environment Agency (EA), 2007. The Quality Protocol for the Production and Use of Quality Compost from Source-Segregated Biodegradable Waste. Environment Agency, Bristol Available from: 〈http://www.environmentagency.gov.uk/commondata/acrobat/compostqp_1721787.pdf〉.

Kay, D., Wyer, M.D., Crowther, J., Stapleton, C., Bradford, M., McDonald, A.T., Greaves, J., Francis, C., Watkins, J., 2005. Predicting faecal indicator fluxes using digital land use data in the UK's sentinel Water Framework Directive catchment: the Ribble study. Wat. Res., 39, 3967–3981.

Kay, D., Aitken, M., Crowther, J., Dickson, I., Edwards, A.C., Francis, C., Hopkins, M., Jeffrey, W., Kay, C., McDonald, A.T., McDonald, D., Stapleton, C.M., Watkins, J., Wilkinson, J., Wyer, M., 2007. Reducing fluxes of faecal indicator compliance parameters to bathing waters from diffuse agricultural sources, the Brighouse Bay study, Scotland. Environ. Pollut. 147, 138–149.

Kay, D., Stapleton, C.M., Crowther, J., Wyer, M.D., Fewtrell, L., Edwards, A., McDonald, A.T., Watkins, J., Francis, F., Wilkinson, J., Faecal indicator organism compliance parameter concentrations in sewage and treated effluents. Wat. Res., in press.

Murphy, J., Riley, J.P., 1962. A modified single solution method for the determination of phosphate in natural waters. Anal. Chim. Acta 27, 31–36.

Rowland, A.P., Haygarth, P.M., 1997. Determination of total dissolved phosphorus in soil solutions. J. Environ. Qual. 26, 410–415.

SCA, 2000. The Microbiology of Recreational and Environmental Waters: Methods for the Examination of Waters and Associated Materials. Standing Committee of Analysts, Environment Agency.

SCA, 2002. The Microbiology of Drinking Water: Methods for the Examination of Waters and Associated Materials. Standing Committee of Analysts, Environment Agency.

Scottish Executive, 2005. Prevention of Environmental Pollution from Agricultural Activity. Available from: 〈http://www.scotland.gov.uk/library5/environment/pepfaa.pdf〉 (accessed 03.01.06).

Searle, P.L., 1984. The Berthelot or indophenol reaction and its use in the analysis chemistry of nitrogen. Analyst 109, 549–565.

SEPA, 2003. Woodchip Corrals. In: Pollution Review, vol. 16. Scottish Agricultural Pollution Group, pp. 6–7. Available from: 〈http://www.sepa.org.uk/pdf/publications/leaflets/sapg/reviews/2003.pdf〉.

Stapleton, C.M., Wyer, M.D., Crowther, J., McDonald, A.T., Kay, D., Greaves, J., Wither, A., Watkins, J., Francis, C., Humphrey, N., Bradford, M., 2007. Quantitative catchment profiling to apportion faecal indicator organism budgets for the Ribble system, the UK's sentinel drainage basin for Water Framework Directive research. J. Environ. Manage. 87, 535–550.

Vinten, A.J.A., Donnelly, S., Ball, B.C., Crawford, C.E., Richie, R.M., Parker, J.P., 2006. A field trial to evaluate the pollution potential to ground and surface waters from woodchip corrals for over-wintering livestock outdoors. Soil Use Manag. 22, 82–94.

ELSEVIER

Journal of Environmental Management 87 (2008) 591–599

Journal of
Environmental
Management

www.elsevier.com/locate/jenvman

Farmyard point discharges and their influence on nutrient and labile carbon dynamics in a second order stream draining through a dairy unit

A.C. Edwards[a], P.S. Hooda[b,*]

[a]*Nether Backhill, Ardallie, By Peterhead AB42 5BQ, UK*
[b]*Centre for Earth and Environmental Science Research, School of Earth Sciences and Geography, Kingston University, Kingston upon Thames, Surrey KT1 2EE, UK*

Received 28 September 2006; received in revised form 11 February 2007; accepted 19 February 2007

Abstract

Two small piped sources deriving from a single farmyard together with the receiving second order stream above and below the farmyard region were sampled over a two-year period. Although not measured directly, observations at the time of sampling suggested that maximum drain flow was about 2% of downstream base flow. Both point sources were flowing on each sampling occasion (~62) and usually had concentrations of phosphorus (P), nitrate (NO_3–N) and biological oxygen demand (BOD) well above those from the upstream site. Individual sample concentrations ranged over more than two orders of magnitude for most determinants and a large proportion of the total P was present as soluble (inorganic and organic) and therefore labile forms. More than 70% of samples collected at the downstream site had concentrations that were >1.2 times those of the corresponding upstream site. On certain sampling occasions $>80\%$ of total dissolved phosphorus (TDP) and $>90\%$ of the BOD and NO_3 instantaneous load appeared to originate from the farmyard region with the composition of downstream samples being completely overwhelmed after the passage through the farmyard. Extrapolations using instantaneous loads suggest that the farmyard and adjacent areas contributed on average 25–30% of the total and dissolved annual downstream P load of $3\,\mathrm{kg\,P\,ha^{-1}}$ and $1.7\,\mathrm{kg\,P\,ha^{-1}}$, respectively. There was no clear relationship between the relative proportion of the contaminant loading originating from the farmyard region and hydrological events. This emphasises the potential localised significance that small, highly concentrated, continuous or semi-continuous farmyard sources can impact headwater streams during periods of low stream flow.

Keywords: Phosphorus; Nitrate; BOD; Farmyard; Point source; Dairy washings; Headwater

1. Introduction

The majority of agriculturally related contaminants that are delivered to surface waters are often perceived to derive from diffuse sources. This general assumption is evident from a number of recent reviews and forms the underlying basis for the application and use of loss coefficients (Hooda et al., 2000a; Vanni et al., 2001; Woli et al., 2002; Salvia-Castellvi et al., 2005). Recent evidence suggests that under certain circumstances, point sources, which include piped flows from farmyards, silage effluents and domestic septic systems (Hooda et al., 2000b), together with surface runoff from areas of hardstanding (Hively et al., 2005) may also contribute significantly. While these various sources contribute cumulatively to the total catchment export loading there is also the possibility for impacts, including increases in biochemical oxygen demand and decline in benthic macroinvertebrate species, on receiving headwaters (Hooda et al., 2000b).

Farmyard areas present a number of potential transport routes that include a combination of surface generated runoff from areas of hardstanding and small point type sources. Contrasts in the extent to which individual sources rely upon precipitation events for their mobilisation and delivery attributes changes the dominance of individual transport pathways.

---

*Corresponding author.
E-mail addresses: t.edwards25@btinternet.com (A.C. Edwards), p.hooda@kingston.ac.uk (P.S. Hooda).

0301-4797/$ - see front matter © 2007 Elsevier Ltd. All rights reserved.
doi:10.1016/j.jenvman.2007.02.013

The low permeability of hardstanding areas increases their susceptibility and capacity for generating large volumes of rapid runoff during storm events. The continuous and direct delivery of point sources into adjacent waterways during periods of low stream flow implies that local water quality issues may well develop (Nagumo et al., 2004; Bowes et al., 2005). The greater temporal and spatial heterogeneity of pollutant delivery to surface waters is therefore increasingly being recognised.

Point sources are often associated with drains from farm buildings which offer a range of potential pollutants that include nutrients, labile organic matter and faecal pathogens. The relative composition can vary widely in relation to farm enterprise and particular operational management practices. The prevalence of surfaces having low permeability coupled with renewable sources of contamination makes farmyards a high potential risk category and the few studies that have quantified losses suggest that this is the actual situation (Edwards et al., this issue). Here we describe the results of regular sampling of two point source discharges that arise from different parts of a farmyard located on a dairy enterprise in SW Scotland. Comparison of stream samples, collected above (upstream) and below (downstream) of the farmyard and point sources, is also used to provide an indication of the extent of local pollution with respect to nutrients and labile organic compounds (BOD).

2. Material and methods

2.1. Site characteristic

The Caddell catchment is located in Ayrshire, SW Scotland, an intensive dairy farming region and lies in a subcatchment of the River Garnock. The major land use is maintained grassland with rough grazing in the upper reaches. Swards are rotationally grazed by cattle and also cut for silage; they are grazed by sheep during the winter months when cattle are housed. The inputs of fertilizer N and P in this region varied between 200–300 kg N $\text{ha}^{-1}\,\text{y}^{-1}$ and 9–22 kg P $\text{ha}^{-1}\,\text{y}^{-1}$, respectively (at the time of sampling), with re-seeded swards generally receiving the largest rates of fertilizer N and P.

The boundary of the farm used for sampling represents only a small part (50 ha) of the whole catchment (705 ha). The farm buildings are situated within 20 m of the stream and slurry is spread on surrounding fields, some of which are adjacent to the stream (Fig. 1). The farmyard covers an area of about 200 m^2. A large slurry store is situated about 5 m from the stream bank, and there was often evidence of slurry residues on the hard floor around the store. This area drains into an open drain that brings farmyard washings, dirty water from the buildings and milk parlour washings into the stream (point source 2; Fig. 1).

2.2. Collection of water samples

The water sampling programme involved taking periodic grab samples from several well-defined locations, including the main channel and two point sources of pollution between the upstream and downstream stations. In the first year (October 1993–September 1994), the stream was sampled at three-weekly intervals and weekly during the second year (October 1994–September 1995). Point source 1 (PS 1) drains an enclosed area used for calf rearing while the second point source (PS 2) has a mixed origin which included washings from the milking/parlour area and probably also receives contaminated flow from a slurry

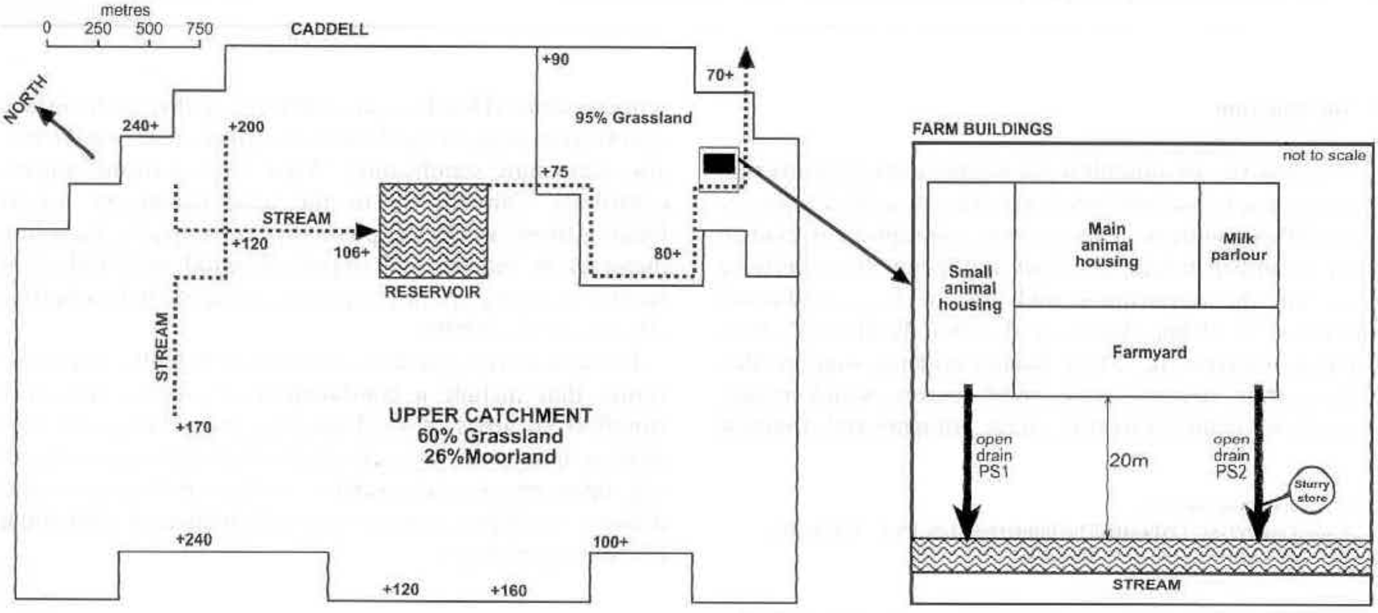
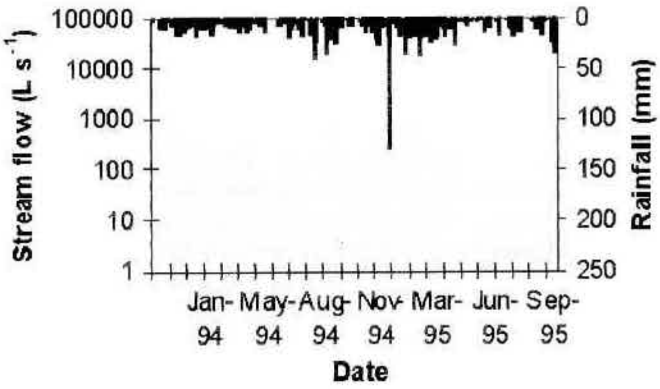


Fig. 1. Cellular diagram of the study catchment, with two point-sourced (PS 1 and PS 2) inputs into the stream.

tore area (Fig. 1). Samples were collected during day time, ften between 10 am and 2 pm. The drainage looked visibly lirty/cloudy when sampled around 10 am compared to amples collected at later times. The drainage from PS 1 ind PS 2 was not metered, however, both discharged highly ariable output. Observations made at the time of sampling howed that combined (PS 1 and PS 2) estimated output aried between approximately 0.5% and 2% of the river ase flow. Clearly there is a certain degree of uncertainty in erms of discharge volumes from the point sources. lowever, it is not likely to introduce noticeable errors in utrient load estimations, as this is a relatively small raction even at the base flow.

Stream stage-heights were recorded using a pressure ransducer and data logger (Campbell Scientific) at the lownstream site. The flow was estimated using a stage-eight and correlating this with a previously calculated elationship between the stage-height and stream flow as lescribed by Hooda et al. (1997). It is a second order tream with the flow varying from 2 L s^{-1} at base flow to bout 1000 L s^{-1} at flood.

Daily rainfall data was obtained from a nearby weather tation, and long-term annual rainfall average in the atchment is 1700 mm.



ig. 2. Comparison of daily rainfall (mm) shown as black bars and stream low (L s^{-1}) shown as grey bars.

2.3. Analysis of water samples

Water samples were filtered through 0.45 μm filters (Milli-pore HVLP) on the day of sampling, and both filtered and unfiltered samples were refrigerated at $<5\,^{\circ}C$ prior to analysis. The filtered samples were analysed for molybdate reactive phosphate (MRP) by the ascorbic acid method (Greenberg et al., 1992) within 24 h of their collection. The filtered and unfiltered water samples were digested in concentrated H_2SO_4 before being analysed for total P by inductively coupled plasma atomic emission spectroscopy (ICP-AES). The P concentration of acid-digested unfiltered water samples represents total phosphorus (TP) and that of filtered samples is total dissolved phosphorus (TDP); particulate phosphorus (PP) was assumed to be the difference between TP and TDP while dissolved organic P (DOP) was assumed to be the difference between TDP and MRP.

The filtered samples were analysed colorimetrically (Jackson, 1958) for nitrate within 24 h of their collection. Unfiltered water samples were tested for the 5-day biochemical oxygen demand (BOD_5) (Anon, 1988).

3. Results

3.1. Hydrological conditions

Rainfall measured for the sampling period from 20/10/93 to 30/9/95 was 3344 mm. The pattern of daily rainfall (Fig. 2) followed the general seasonal trend typical of the region with a drier summer period and a wetter spring and autumn. This pattern was evident in stream flow which varied over two orders of magnitude. In one extreme precipitation event during the autumn of 1993, a daily rainfall totalled >100 mm. A regular weekly (or longer) stream sampling programme is generally biased towards collection under low flow periods. Comparison of flow duration curves for the whole sampling period (Fig. 3a) against the actual flow on the sampling days (Fig. 3b) shows a similar general trend although the extreme event where stream flows of over 1000 L s^{-1} occurred for ~1% of

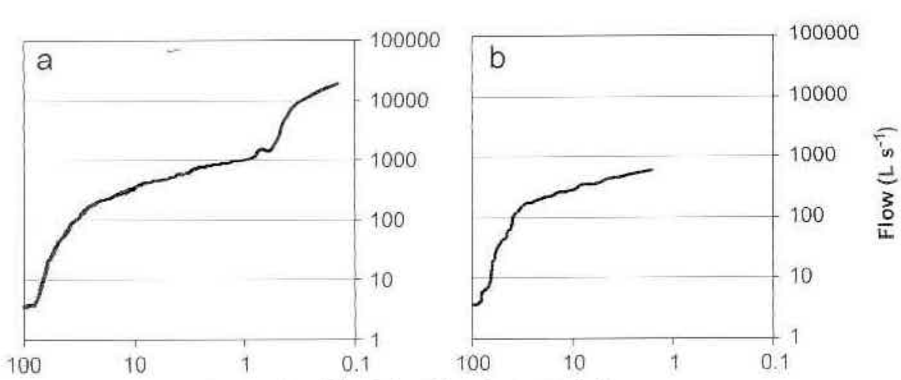


Fig. 3. Flow duration curve of the downstream site over the sampling period (a) and on the actual day of sampling (b), based upon average daily flows.

Table 1
Range in composition ($mg\,L^{-1}$) of two point sources (PS 1 and PS 2) sampled periodically over a two-year period

	TP	PP	TDP	MRP	DOP	BOD	NO_3–N
Number of samples	63	62	63	68	62	16	68
PS1							
Average	1.12	0.62	0.52	0.26	0.25	33.2	3.28
Min	0.13	<DL	0.06	0.01	<DL	3.24	0.40
Max	5.71	5.49	4.64	4.16	1.08	87.3	18.5
PS2							
Average	5.77	1.81	3.96	2.14	1.74	50.6	2.81
Min	0.28	0.03	0.10	0.05	<DL	6.60	0.45
Max	51.0	16.2	34.8	27.8	19.0	197	14.7

Abbreviations are total P – TP, particle P – PP, total dissolved P – TDP, molybdate reactive P – MRP, dissolved organic P – DOP, biochemical oxygen demand – BOD, nitrate – NO_3–N, DL – detection limit.

the time was not sampled. The similarity in trend between duration curves has been used in a final section that extrapolates instantaneous loadings to provide annual estimates.

3.2. Comparison of point sources

Samples were collected from each point source on each occasion indicating their continuous flow properties. Individual point sources had a variable composition (Table 1), for example, the average TP concentration of the PS 2 was ~5 times greater than PS 1 ($p<0.001$). The various forms of P were all significantly (MRP, DOP and TDP $p<0.001$, PP $p<0.05$) different between sources. On average approximately half the TP in the PS 1 was present as TDP and half of this was MRP. This was in contrast to PS 2 which not only had the higher TP concentration but also the larger proportion as TDP. The highest concentration of TP measured over the sampling period was $51\,mg\,L^{-1}$ at PS 2. Concentrations of BOD and NO_3 were similar (no significant difference) between the two point sources although the range in concentrations between individual sampling dates was still large.

Temporal variability in composition between the two point sources suggested the existence of some seasonal 'management' or 'transport' factor influencing concentrations (Fig. 4). While PS 1 tended to be more variable with highest TP concentrations during the autumn, PS 2 by contrast had highest TDP and TP concentrations during the spring and summer with comparatively small winter concentrations. Differences in BOD were less apparent between sources, however, the number of samples were relatively small. Seasonal differences in NO_3–N concentrations were less obvious but were slightly higher for PS 1 during the autumn and winter.

The lack of any clear relationship between individual pollutant groups suggests that each point source has a complex and probably mixed origin. Differences in behaviour and source of individual constituents from each PS are supported by the variable pattern of their correlations (Table 2). While the soluble and particulate P fractions were well correlated the only other correlation for PS 1 was between MRP and BOD ($p<0.05$). The general situation was similar for PS 2 although correlations between the various P fractions and BOD were also highly significant. Interestingly NO_3–N was not correlated with any of the other determinants.

3.3. River concentrations

On most sampling occasions concentrations were greater at the downstream sampling site, as indicated in Fig. 5 by points falling below the 1:1 (unity) line. The magnitude of any difference varied greatly with some samples showing large differences, and this did not appear to be related to the initial upstream concentration. Approximately 80% of MRP concentrations (data not shown) showed a downstream/upstream ratio greater than 1.2 while the corresponding figure for TDP and TP was smaller (~60%). For BOD the majority of samples (>80%) had a ratio greater than 1.2 while the comparable value for NO_3–N was 70% (Fig. 5). Generally the proportion of MRP that constituted TDP and the TDP fraction of the TP both increased as the concentrations of TDP and TP increased (Fig. 6). This suggests that when concentrations of TP increase then the bioavailability is also greater.

The majority of samples from each point source had concentrations of P, BOD and NO_3–N that were greater than those of the receiving water, and this was particularly true for PS 2. The average concentrations of all P fractions, except DOP, were therefore significantly greater at the downstream sampling site (Table 3). On average more than half of the TP was present as TDP of which <50% was MRP. The measured increase in TP concentration was attributable to an increase in both particulate and soluble P fractions. Concentrations of NO_3–N and BOD were also significantly greater at the downstream sampling location. The increase in average concentration, expressed as a ratio for downstream samples, varied but all were greater than 1.5, with the largest proportional increase being found for NO_3 and BOD.

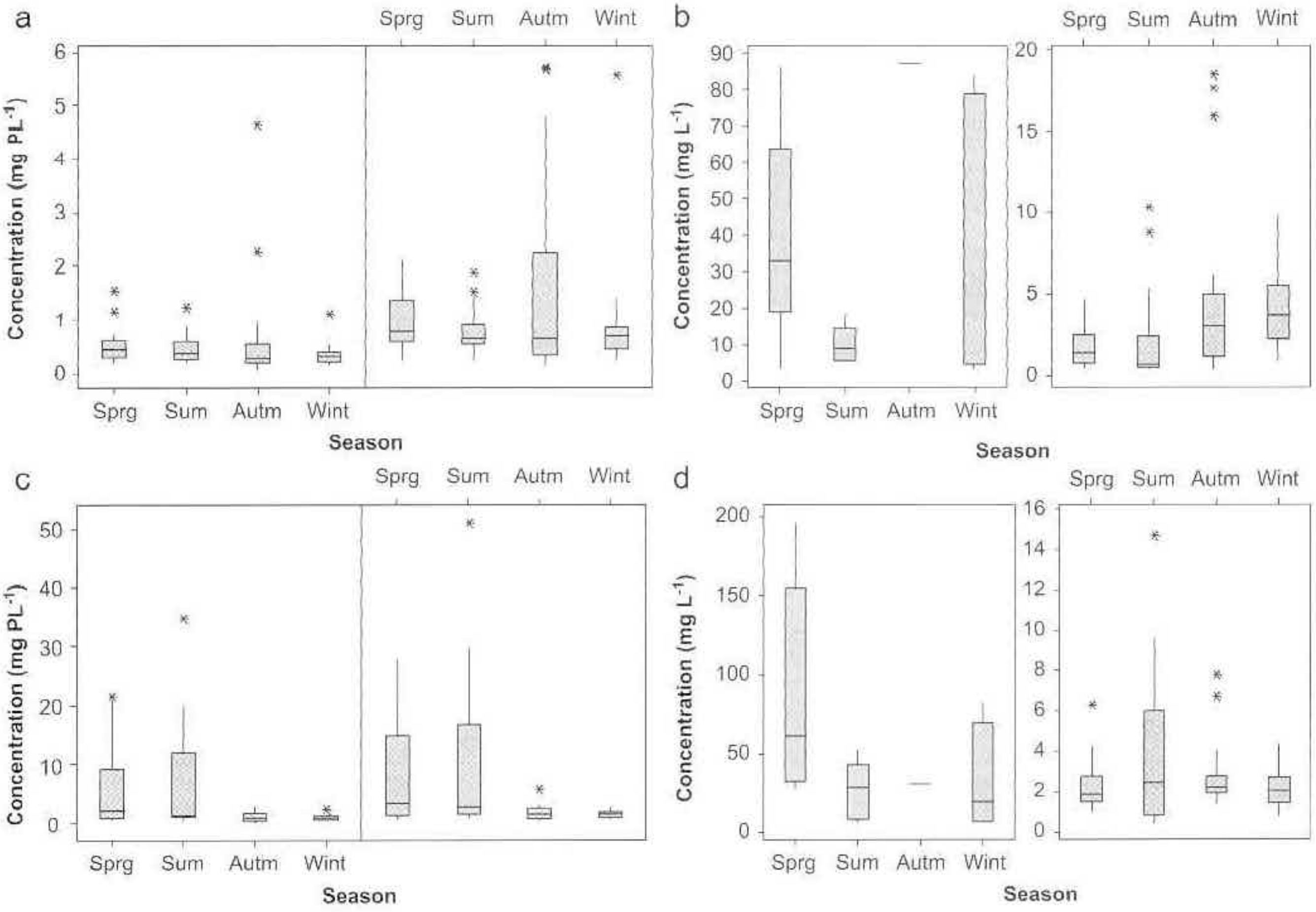


Fig. 4. Seasonal comparison of the concentrations of total dissolved P (TDP) and total P (TP) (a) biochemical oxygen demand (BOD) and NO_3–N (b) for point source 1 and TDP and TP (c) BOD and NO_3–N (d) for point source 2. Where the box represents roughly the middle 50% (interquartile range) of the data, and lines (whiskers) indicate the general extent of the data. The horizontal line inside the box represents the median and the outliers are marked as an asterisk (only one data point was available for the autumn period for BOD PS 1 and PS 2).

Table 2
Correlation betweenvarious determinants present in point sources PS 1 and PS 2

	MRP	TDP	TP	BOD
PS 1				
TDP	***			
TP	***	***		
BOD	*	ns	ns	
NO_3–N	ns	ns	ns	ns
PS 2				
TDP	***			
TP	***	***		
BOD	***	***	***	
NO_3–N	ns	ns	ns	ns

Abbreviations are total P – TP, total dissolved P – TDP, molybdate reactive P – MRP, biochemical oxygen demand – BOD, nitrate – NO_3 N. $^{***}p<0.001$, $^{*}p<0.05$, ns – Not significant.

3.4. Instantaneous flux

A wide range (greater than three orders of magnitude) of instantaneous loads were calculated for the downstream site (Table 4). The proportion of the total downstream load that was attributable to the upstream sources generally averaged between 60% and 75%. This suggests that despite the lack of volumetric flow data for the two point sources and the upstream station the 'farmyard region' contributed on average 25–40% of the total pollutant load. On certain sampling occasions >80% of TDP and >90% of the BOD and NO_3 load appeared to originate from the farmyard region. The instantaneous load measured at upstream and downstream locations is shown in Fig. 7 and ranged over four orders of magnitude. Large differences in instantaneous loadings were apparent for individual sampling dates even at low river flows, supporting the continuous and rainfall independent nature of delivery from the farmyard area. Stream flow was the primary factor responsible for generating these differences and consequently significant positive linear relationships were evident between instantaneous load and stream flow. The individual relationships have been used to calculate daily loads at the downstream site and the difference between upstream and downstream sites. Expressed on an annual basis, total losses at the downstream site were 3 kg P ha^{-1} TP and 1.7 kg P ha^{-1} TDP. Assuming that the difference between upstream and downstream daily loadings relates to those from the

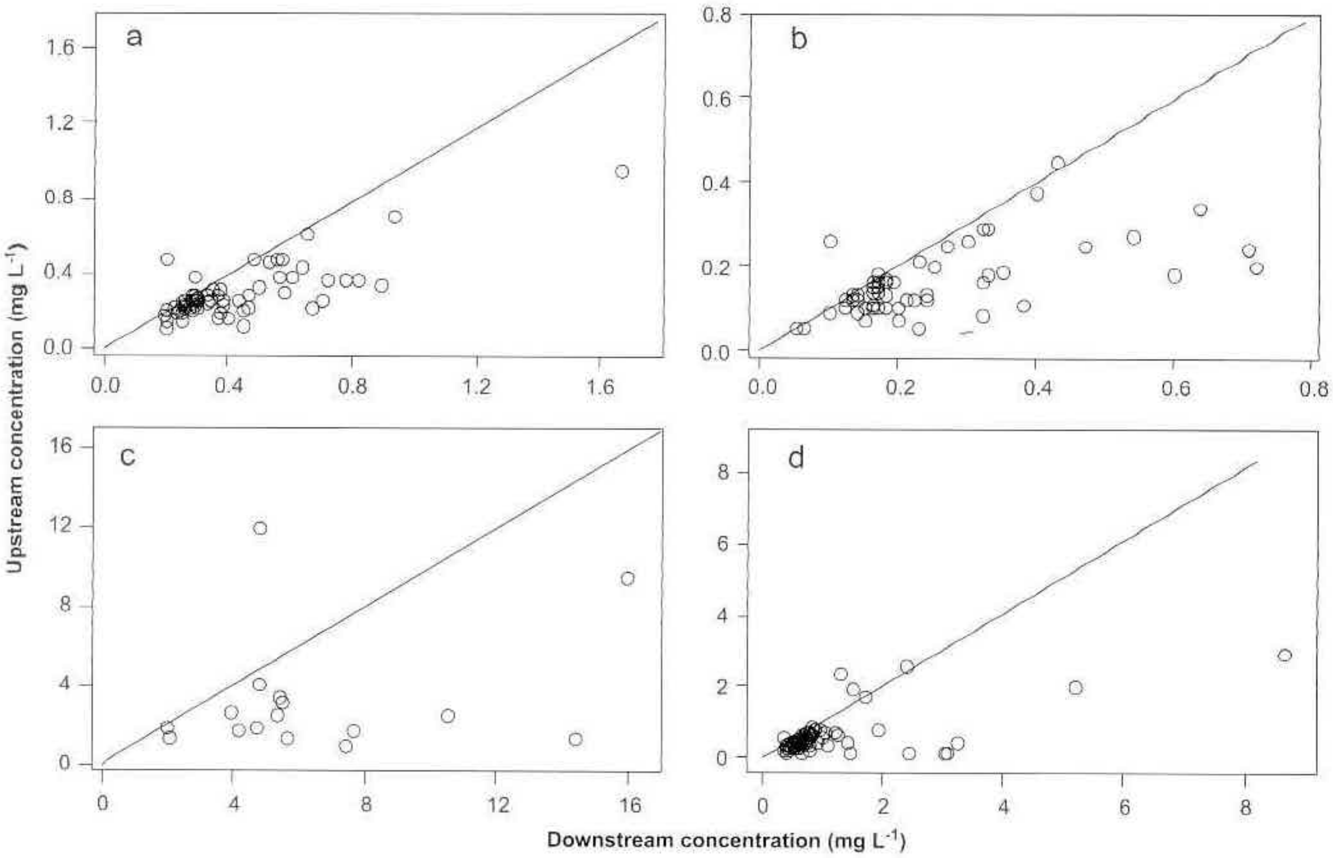


Fig. 5. Comparisons of individual paired sampling of upstream and downstream sites for (a): total P, (b): total dissolved P, (c): biochemical oxygen demand and (d): NO_3–N.

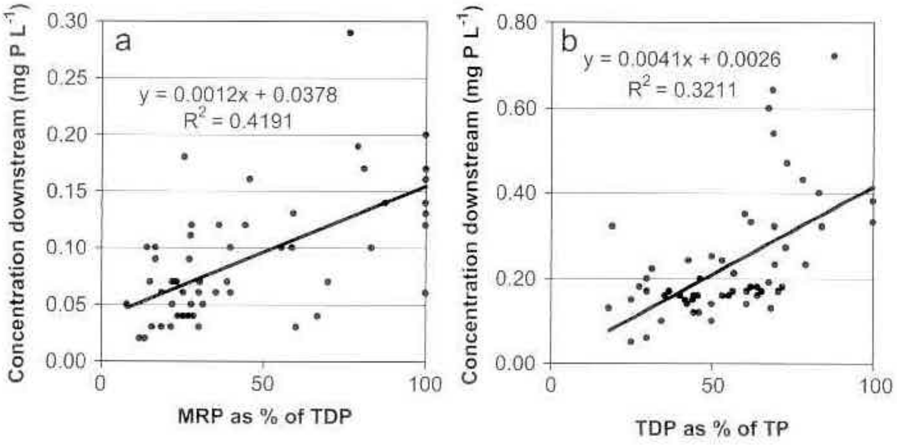


Fig. 6. Relationship between concentrations of (a): molybdate reactive P (MRP) and the proportion (%) of MRP as a fraction of total dissolved P (TDP) and (b): the proportion (%) of TDP as a fraction of total P (TP) for the downstream site. Both significant $r^2 = 0.42$ and $r^2 = 0.32$ ($p < 0.001$).

farmyard region then these accounted for ~30% and 17% of the annual TP and TDP load, respectively.

The role of any seasonal or climatic related factors responsible for mobilising and transporting effluent from point sources is difficult to establish (Fig. 8). The estimated contributions of TP, TDP or NO_3 derived from farmyard sources appeared to display little relationship to stream flow suggesting that these sources could degrade water quality without the need of a major hydrological event.

4. Discussion

Regular sampling and analysis of two point sources derived from a farmyard have demonstrated a variable but often high degree of contamination with respect to nutrients and labile organic matter. Particular emphasis was placed upon quantifying their significance as possible sources of P. This was clearly evident because on >95% of all sampling occasions, one or both of the point sources

Table 3
Comparison of averaged concentrations (mg L^{-1}) for upstream and downstream sites

	TP	PP	TDP	MRP	DOP	BOD	NO_3–N
Number	62	61	62	66	59	16	68
Average up[a]	0.29	0.14	0.16(55)	0.05(36)	0.11	3.28	0.63
Average down[b]	0.43	0.20	0.24(55)	0.10(47)	0.15	6.50	1.07
Significance level	***	*	***	***	ns	**	*
Average ratio[c]	1.50	2.12	1.59	2.11	1.75	3.04	2.69
Average difference	0.12	0.06	0.07	0.04	0.04	3.22	0.43

The proportions (%) of MRP and TDP as a fraction of TDP and TP are shown in brackets. Where abbreviations are total P – TP, particle P – PP, total dissolved P – TDP, molybdate reactive P – MRP, dissolved organic P – DOP, biochemical oxygen demand – BOD, nitrate – NO_3–N. Probability levels *, **, ***: $p<0.05$, 0.01 and 0.001, respectively, ns – not significant.

[a] Average of upstream samples.

[b] Average of downstream samples.

[c] Average of all downstream/upstream ratios.

Table 4
Instantaneous load of various P fractions, BOD and NO_3–N (all mg s^{-1}) at the downstream site together with the averaged proportional (%) contribution derived from upstream sources together with the minimum proportion shown in brackets

Variable	No	Mean (SE)	Minimum	Median	Maximum	Proportion (%)
MRP	62	9.59 (1.92)	0.17	3.65	93.6	61 (8)
TDP	58	24.7 (5.8)	0.58	6.71	272	72 (21)
TP	58	40.8 (8.9)	0.82	12.8	394	72 (26)
BOD	16	822 (236)	19.3	380	2854	60 (9)
NO_3–N	64	91.3 (15.3)	1.13	33.0	622	73 (4)

Abbreviations are total P – TP, total dissolved P – TDP, molybdate reactive P – MRP, biochemical oxygen demand – BOD, nitrate – NO_3–N.

had TP concentrations that were well above those of the adjacent receiving stream water. A similar situation existed for labile organic compounds and therefore also BOD (and presumably faecal indicator organisms) and while the only N form measured was nitrate, concentrations were also high relative to those at the upstream station. It would be expected that a much greater proportion of N was present as reduced forms in effluent draining from farmyards having livestock. Similarly, farmyards have been shown to represent an important source of microbial contamination (Kay et al., 2003). There are direct local influences of farmyards on stream water quality (see Hooda et al., 2000b).

Runoff from impervious hardstanding areas and farm tracks often displays a marked episodic response in relation to hydro-logical events with the degree of contamination dependent upon various combinations of factors (Edwards et al., this issue). Importantly, most farmyards represent a renewable source of potential contaminants, especially where livestock are present.

The highly variable composition of hardstanding runoff, noted previously (Edwards et al., this issue), was also apparent in these two point sources. Both short (daily) and longer-term (seasonal) differences in composition were measured. Some underlying seasonality in composition was apparent but this differed between sources. A combination of management and climatic factors could be potentially responsible for introducing this variability although it was difficult to establish any clear relationship between effluent concentration and rainfall. Point source 2 might be expected to show a highly variable daily concentration probably due to the periodic contribution from dairy parlour washings. In contrast PS 1 might be influenced by more management related factors, such as number and age of housed cattle, and cleaning operations. The changes in concentration of individual components appeared to occur independently of each other which suggested a complex and varying range of contributing contaminant sources in each point source. For example, PS 2, a mixed source, would be expected to receive periodic (twice daily) dairy washings having a typically high BOD concentration relative to their nutrient content, in contrast to washings of urine and faecal material from collecting/feeding areas (Hooda et al., 2000a, b). Fresh animal faeces generally contain the majority of N in reduced forms and only a small proportion of NO_3–N.

It is likely that with this general geographical region, where livestock enterprises are the predominant farm type, individual stream passess through a number of similar situations where farmyard areas might contribute to contaminant loads. Estimating the contribution individual farmyards make to the total loading is difficult due to selective retention (e.g., sedimentation of PP) and transformation (e.g., de-nitrification or sorption of MRP)

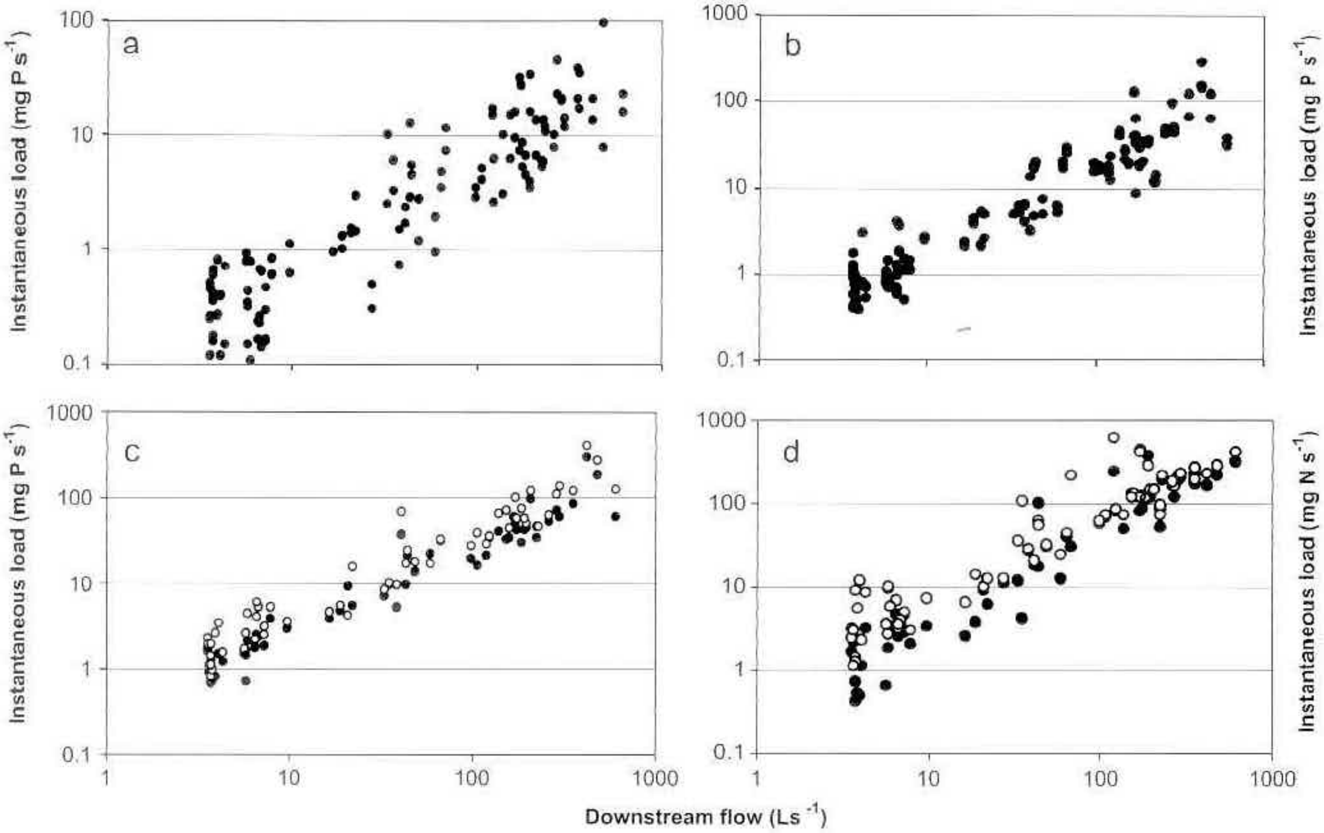


Fig. 7. Comparison of upstream and downstream instantaneous load (mg s^{-1}) of various P fractions (a): Molybdate reactive, (b): total dissolved, (c): total and (d): NO_3–N plotted against downstream flow (note logarithmic scales).

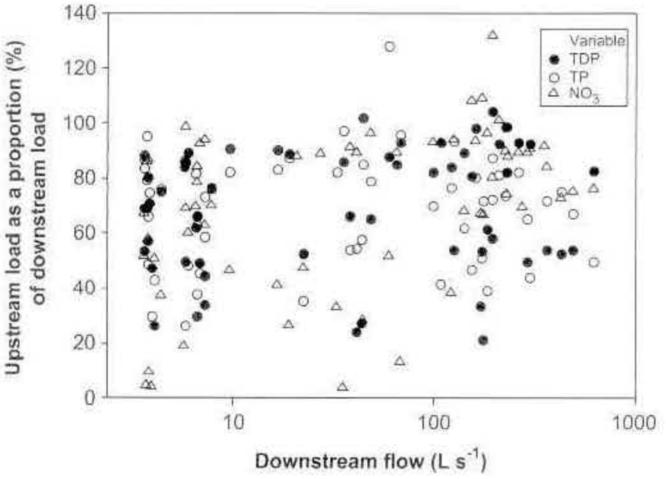


Fig. 8. Proportion (%) of instantaneous downstream load accounted for by the upstream load of total dissolved P, total P and NO_3 expressed against down-stream discharge (one pair of data were omitted as obvious outliers).

processes that can occur downstream. A combination of geographical and seasonal differences in annual rainfall–runoff relationships will influence the volume of runoff generated in addition to the resulting within-stream dilution. The overwhelming influence that discharge has on the calculation of substance loads resulted in a positive relationship between instantaneous load and stream flow. The reasonable relationship between average daily flow conditions and flow at the time of sampling meant that a tentative extrapolation to provide estimates of annual P loads and the proportional contribution from the farmyard region was possible. An estimated 30% and 17% of the annual TP and TDP load for this 700 ha catchment was derived from the farmyard region.

5. Conclusions

Two small point source drains from a dairy farmyard area were continually discharging contaminated effluent into a small stream system. The point sources were continuous in their flow properties and displayed a variable composition. On most sampling occasions one or more of the point sources experienced concentrations of P and BOD that were well above those of the receiving stream water. This single farmyard contributed a significant proportion (30%) of the annual total load of P that was calculated to be derived from this 700 ha catchment.

Acknowledgments

This work was funded by the Scottish Executive Environmental Research and Agriculture Department through a joint grant to Scottish Agricultural College and Macaulay Institute. The referees are thanked for their useful comments.

References

Anon, 1988. Methods for the Examination of Waters and Associated Materials; 5-Day Biochemical Oxygen Demand, second ed. HMSO, London, pp. 3–35.

Bowes, M.J., Hilton, J., Irons, G.P., Hornby, D.D., 2005. The relative contribution of sewage and diffuse phosphorus sources in the river Avon catchment, southern England: implications for nutrient management. Science of the Total Environment 344, 67–81.

Edwards, A.C., Kay, D., McDonald, A., Francis, C., Watkins, J., Wilkinson, J.R., Wyer, M.D., 2008. Farmyards, an overlooked source for highly contaminated runoff. Journal of Environmental Management 87, 551–559.

Greenberg, A.E., Clesceri, L.S., Eaton, A.D. (Eds.), 1992. Standard Methods for the Examination of Water and Wastewater. EPS Group Inc., Hanover, Maryland, USA.

Hively, W.D., Bryant, R.B., Fahey, T.J., 2005. Phosphorus concentrations in overland flow from diverse locations on a New York dairy farm. Journal of Environmental Quality 34, 1224–1233.

Hooda, P.S., Edwards, A.C., Anderson, H.A., Miller, A., 2000a. A review of catchment water quality concerns in livestock farming areas. Science of the Total Environment 250, 143–167.

Hooda, P.S., Moynagh, M., Svoboda, I.F., Thurlow, M., Stewart, M., Thomson, M., Anderson, H.A., 1997. Streamwater nitrate concentrations in six agricultural catchments in Scotland. Science of the Total Environment 201, 63–78.

Hooda, P.S., Moynagh, M., Svoboda, I., Miller, A., 2000b. Macroinvertebrate as bioindicators of water pollution in streams draining dairy farm catchments. Chemistry and Ecology 17, 17–30.

Jackson, M.L., 1958. Soil Chemical Analysis. Prentice-Hall, New Jersey, 498 pp.

Kay, D., Edwards, A.C., Ferrier, R.C., Francis, C., Kay, C., Rushby, L., McDonald, A.T., Crowther, J., 2003. Does farmyard runoff represent a significant source of faecal indicator organisms for the bathing waters of SW Scotland? In: Proceedings of the 7th International Water Association Specialised Conference on Diffuse Pollution and Basin Management, vol. 3, Dublin, pp. 22–29.

Nagumo, T., Woli, K.P., Hatano, R., 2004. Evaluation of the contribution of point and non-point sources of nitrogen pollution in stream water in a rural area of central Hokkaido, Japan. Soil Science and Plant Nutrition 50, 109–117.

Salvia-Castellvi, M., Iffly, J.F., Borght, P.V., Hoffman, L., 2005. Dissolved and particulate nutrient export from rural catchments: a case study from Luxembourg. Science of the Total Environment 344, 51–65.

Vanni, M.J., Renwick, W.H., Headworth, J.L., Auch, J.D., Schaus, M.H., 2001. Dissolved and particulate nutrient flux from three adjacent agricultural watersheds: a five-year study. Biogeochemistry 54, 85–114.

Woli, K.P., Nagumo, T., Hatano, R., 2002. Magnitude of nitrogen pollution in stream water due to intensive livestock farming practices. Soil Science and Plant Nutrition 48, 883–887.

ELSEVIER

Journal of Environmental Management 87 (2008) 600–608

Journal of Environmental Management

www.elsevier.com/locate/jenvman

The microbial status of natural waters in a protected wilderness area

A.T. McDonald*, P.J. Chapman, K. Fukasawa[1]

Earth and Biosphere Institute, School of Geography, University of Leeds, Leeds, LS2 9JT, UK

Abstract

Waters derived from remote 'wilderness' locations have been assumed to be largely free of bacterial contamination and thus such, near-pristine, protected catchments, unused for agriculture, have been first in the multiple line of protection (pristine catchment—long storage—treatment—disinfection) employed by the water industry. This assumption is challenged by a bacterial survey of the waters derived from the New Cairngorm National Park, Scotland. Over 480 spot samples were taken for 59 sites between March 2001 and October 2002 during nine field campaigns each of three to five days duration. Over 75% of samples tested positive for *Escherichia coli* (*E. coli*) and 85% for total coliforms. Concentrations displayed both temporal and spatial patterns. Largest values occurred over the summer months and particularly at weekends at sites frequented by visitors, either for 'wild' camping or day visits, or where water was drawn from the river for drinking. Overall the spatial and temporal variations in bacterial concentrations suggest a relationship with visitor numbers and in particular wild camping. The implications of the results for drinking water quality and visitors health are discussed along with possible management options for the area in terms of improving the disposal of human waste.

Keywords: Wilderness; Total coliforms; *E. coli*; Wild camping; Human waste

1. Introduction

The World Conservation Union, defined a 'protected area' as an area of land and/or sea especially dedicated to the protection and maintenance of biological diversity, and of natural and associated cultural resources, and managed through legal or other effective means (IUCN, 1994). While the conditions of establishment of protected areas vary greatly from one country to another depending on needs, priorities, and on differences in legislative and financial support, protected areas, such as wilderness areas and national parks, are generally managed with two main aims in view, that is, (i) conserving the special qualities of the area and (ii) providing recreational experiences. These attributes, special qualities creating recreational potential, however, inevitably attract to these areas significant visitor numbers. Since 1965, recreational use of wilderness areas in the US has grown by nearly 400% (Hampton and Cole, 1995), increasing significantly during the 1990s and likely to intensify in the future (Cole, 1996). A survey of all national parks in the UK, US, and Japan indicated that such upwards trends in demand were a continuing and global phenomena (Fukasawa, 2004). This increase in visitor numbers to protected areas presents managing agencies with a balancing act of protecting vulnerable areas and resources from impacts associated with large numbers of visitors whilst permitting access and maintaining safety.

Whilst great attention has been paid to the issues of trampling of vegetation, erosion of soil on footpaths (for example, Watanabe and Fukasawa, 1998; Hampton and Cole, 1995; Cole, 1991, 1983) and disturbance of wildlife (for example, Hendee et al., 1990), the issue of human waste disposal and potential for ground and surface water pollution has been largely ignored. Facilities for walkers or campers (e.g. toilets and drinking water) are not normally provided in protected areas, such as national parks, and indeed there are cogent and persuasive arguments to support that non-provision. However, how to deal with the increasing amount of waste produced by visitors is a

---

*Corresponding author. Tel.: +44 113 343 3344; fax: +44 113 343 3308.
E-mail address: a.t.mcdonald@leeds.ac.uk (A.T. McDonald).

[1] Current address: Halcrow Group Limited, Burderop Park, Swindon SN4 0QD, UK.

0301-4797/$ - see front matter © 2007 Elsevier Ltd. All rights reserved.
doi:10.1016/j.jenvman.2007.10.007

non-trivial disposal problem and is an important management issue for every protected area around the world. Currently there is little applied research on the impacts of human waste disposal on the environment in wilderness areas or on the quality of streamwater in such sites (Cilimburg et al., 2000). Most wilderness areas provide information to visitors on best practices for disposal of human faeces, the most common methods of which are (i) shallow burial in soil, (ii) latrines, (iii) surface disposal and (vi) carrying out of faeces. In most cases these suggestions are based on observations and experience instead of research and the effectiveness of each method is likely to vary between areas depending on climate and soil characteristics.

The improper disposal of human faeces presents two major concerns. Firstly, human health problems relating to the transmission of disease-causing pathogens (bacteria, viruses and protozoan) from human faeces as a consequent of either direct contact or contamination of drinking water and secondly aesthetic concerns of visitors who find improperly disposed human waste. In wilderness areas, surface waters are often used for drinking and cooking and/or recreational use. Hence it is important that these waters are safe for these uses and do not pose a risk to public health.

Bacterial contamination of surface waters in wilderness areas primarily originates in the surface soils that contain background microorganisms and those originating from human, domestic and wild animal faeces (Cole, 1990; Silsbee and Larson, 1982). More water quality studies have been conducted in easily accessible recreation sites or on municipal water reservoirs (McDowell, 1979) often sited in marginal 'wilderness' areas. Overall, these studies have found that bacterial contamination occurs in areas receiving high use at peak periods of time (Kuss et al., 1990). Fewer studies, mainly in the USA, have investigated the bacterial status of remote wilderness waters and any relationship with recreational use (e.g. Silsbee and Larson, 1982; Gary and Adams, 1985; Cole, 1990), and the results of these studies have often been contradictory or inconclusive (Cilimburg et al., 2000; Hammit and Cole, 1998). This may partially reflect the fact that impacts from human waste on water quality are believed to be localized, temporary and dependent on environmental variables (Varness et al., 1978; Kuss et al., 1990) and the inherent difficulties associated with carrying out water quality studies in wilderness areas, such as problems of access and discriminating between background bacterial levels and inputs from non-human sources (Hermann and Williams, 1986). Hence there is a paucity of studies that have investigated the impact of human waste disposal on water quality in wilderness areas, and those that have been carried out are characterised by being of relatively short duration and limited number of samples.

The present study, reported in this paper, was, therefore, undertaken to determine the spatial and temporal distribution of stream water bacterial concentrations in a remote wilderness area in the UK under a variety of both user intensities and hydrological conditions.

2. Materials and methods

The study was carried out within the Mar Lodge Estate in north–east Scotland, approximately 75 km from the North Sea (Fig. 1). The Mar Lodge Estate has been managed by the National Trust for Scotland (NTS) since 1995 and lies within the Cairngorms National Park, established in 2005; an area cited as being the last true wilderness in the UK and holding remnant arctic tundra landscape (Curry-Lindahl, 1990). The estate contains four of the five highest mountains in Britain and remnant Caledonian Pine forest of national importance. The terrain is predominantly moorland with arctic/alpine vegetation on the higher ground and supports low density deer grazing.

Access to the estate by the public is restricted to foot as no public road crosses the estate. A car park is provided at the Linn of Dee (Fig. 1) on the eastern edge of the estate, 5 km west of Braemar. Due to the long distance between the car park and the summits of the high mountains, many walkers and mountaineers carry tents for overnight stays. The most popular area for camping wild is around Derry Lodge, 4.3 km from the Linn of Dee (Fig. 1). There are also a few mountain huts; only Bob Scott's Bothy has a toilet (a hole in ground located 10 m from river). The estate does not condone wild camping, however, the concentration of tents, which can range from one or two to over 100 tents per night, in one popular location around Derry Lodge is causing concern as no sanitary facilities are provided in terms of toilet or drinking water. Therefore, this study centred on Derry Lodge (57°01′ 22″N, 3°34′50W) at the confluence of the Luibeg Burn and the Derry Burn (Fig. 1). The choice of site allowed an evaluation of the bacterial status of a raw water that was used for direct consumption; permitted the tracing of the contamination, if it occurred, between two rivers of differing recreational use (Derry Burn versus Luibeg Burn); and allowed an evaluation of the recovery rate downstream (now called the Lui Water) before its confluence with the much larger River Dee.

The logistics of sampling in a remote location required that the sampling was conducted on a campaign basis with sampling and analysis over a period of three to five consecutive days in each campaign. Nine sampling campaigns were conducted over a period of eighteen months; the first campaign occurred between the 26th and 31st August 2000, seven campaigns were conducted between 28th June 2001 and 31st October 2001, and the final campaign was in late winter between 1st and 6th March 2002.

The design of the field-sampling programme was constrained by two factors. Firstly, the time available to collect water samples each day was limited because each water sample must be analysed within six hours of collection (a limitation that has now been relaxed to

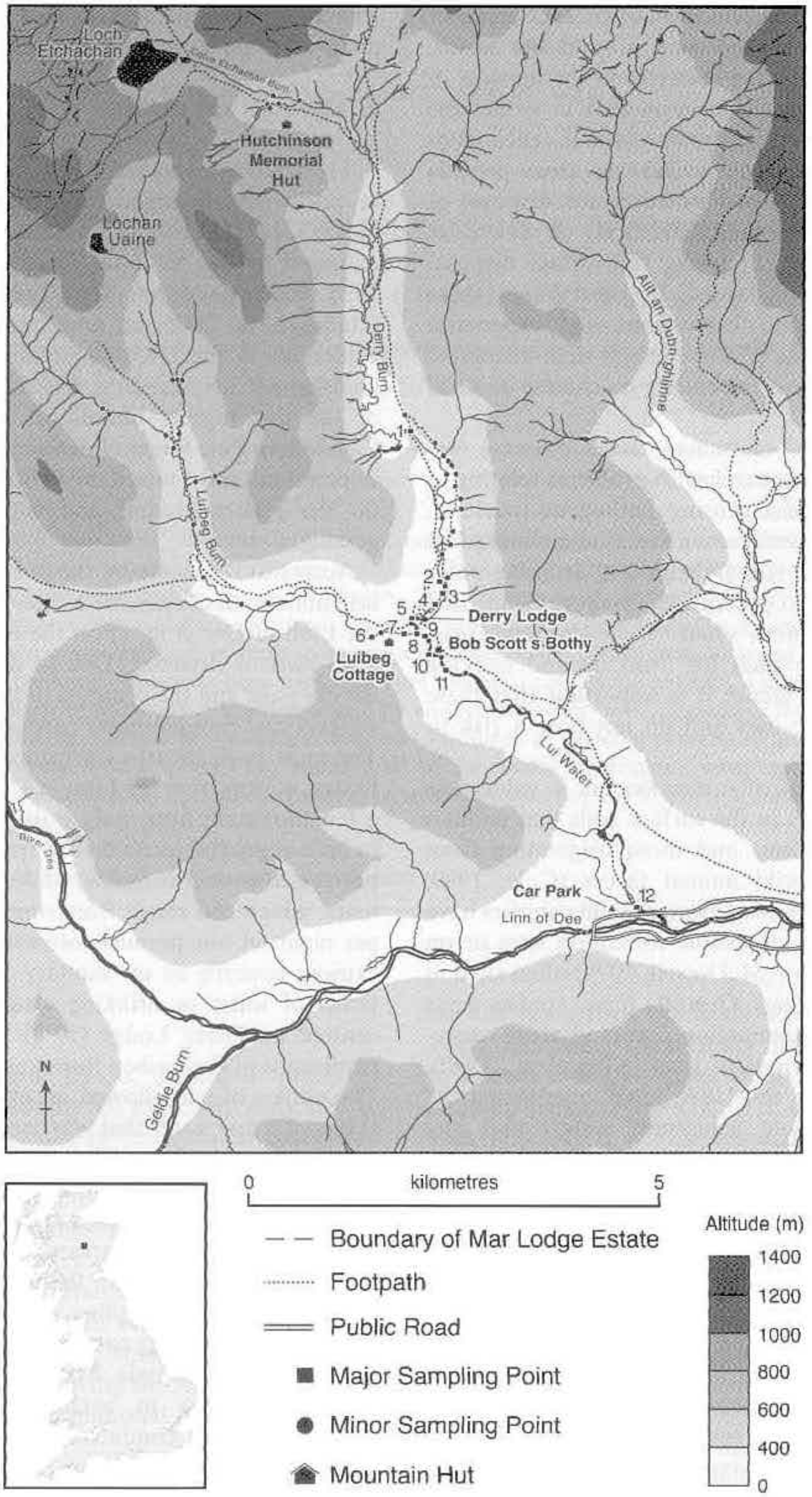


Fig. 1. The study area at Mar Lodge Estate, Scotland, showing the location of the water sampling sites.

24 h). Secondly, vehicle access beyond Derry Lodge was not permitted so all sampling points past this location were reached by foot. These factors restricted (i) the number of samples collected each day and (ii) the spatial coverage of the area around Derry Lodge.

The sampling sites were mainly situated on Derry Burn and Luibeg Burn (Fig. 1). Twenty eight sites were situated along Derry Burn, including 13 tributaries, 18 sites along the Luibeg Burn and 10 sites along the Lui Water, including one tributary site. In total, from 59 sites, 481 samples were collected and analysed. The location of the 59 sampling sites is shown in Fig. 1. Some of these sites represent spot sampling of the remote high altitude waters of the eastern Cairngorm slopes. Regular sampling was focussed on 12 major sampling sites (see Fig. 1) at lower altitude because they were sites that were close to informal 'wild' camping grounds, a mountain hut and long distance footpath intersections. Hence these sites were more likely to be selected for drinking/cooking water.

Stream water samples were taken from the near surface (0–5 cm) using a 500 ml sterile plastic bacteriological bottle from Aurora Scientific®. All bottles contained sodium thiosulphate ($Na_2S_2O_3 \cdot 5H_2O$), which is used to neutralise any chlorine or chloramines that may be present in the water sample. Samples were immediately kept in the dark, cooled in an insulated container and transported to the laboratory within four hours, and processed within six hours, of collection as specified in standard methods (CREH, 1999).

The presence of total coliforms and *Escherichia coli* in the water samples was determined by filtering measured volumes of water, in this study three dilutions; 100, 10 and 1 ml of each sample were filtered, through either cellulose acetate or cellulose nitrate membranes. The membranes were then incubated on Membrane Lactose Glucuronide Agar (m-LGA) contained in sterile 50 mm petri dishes following the methods of Sartory and Watkins (1999) and Watkins and Jian (1997). The media were made up in a sterile laboratory, CREH, in Leeds and were suitable for use for up to seven days when kept cool. Each petri dish was incubated at 30 °C ± 1 °C for 4 h followed by 37 °C ± 1 °C for 14 h. On completion of the full 18 h incubation, all membranes were examined for presence of visible colonies. All yellow colonies were counted as presumptive non *E. coli* and all green colonies counted as presumptive *E. coli*. The sum of the yellow and green colonies gives the number of total coliform in each sample. Plates of m-LGA with counts between 20 and 80 colonies were selected for reporting the results. Results are presented as a logarithmic transformation of the number of colonies present in each sample.

3. Results

3.1. Occurrence and amount of bacteria

Samples were collected from 59 sites (Fig. 1) and total coliform was detected in samples from 49 sites and *E. coli* was detected in samples collected from 47 sites. At all of the sites where total coliform and *E. coli* were absent, only one sample was collected, apart from one site where three samples were collected. In total, 35 sites were only monitored once or twice during the whole study period, most of these were of first or second order, high altitude streams. At 20 sites, more than five samples were collected and all these sites had samples that tested positive for the presence of total coliform and *E. coli*.

In total, 481 stream water samples were collected and analysed for total coliform and *E. coli* during this study; 70% of which were collected from the major sampling sites (Fig. 1). Total coliform was detected in 85% of these samples and *E. coli* in 75%. The majority of samples (56% for total coliform and 52% for *E. coli)* in which bacteria were detected contained between one and nine colonies (Fig. 2). However, 7% of samples contained > 100 total coliform colonies per 100 ml and 3% of samples contained > 100 *E. coli* colonies per 100 ml (Fig. 2).

3.2. Spatial variation in the microbial status of natural waters

There are four main rivers in the study area; these are (i) Derry Burn, (ii) Luibeg Burn, (iii) Lui Water and (iv) River Dee. Each of these rivers experience differing visitor pressures and recreational usage. While campers congregate along Derry Burn, particularly between sites 2 and 5, considerably less wild camping occurs along Luibeg Burn, which is more exposed. The confluence of the Derry and Luibeg Burns forms Lui Water which joins the River Dee

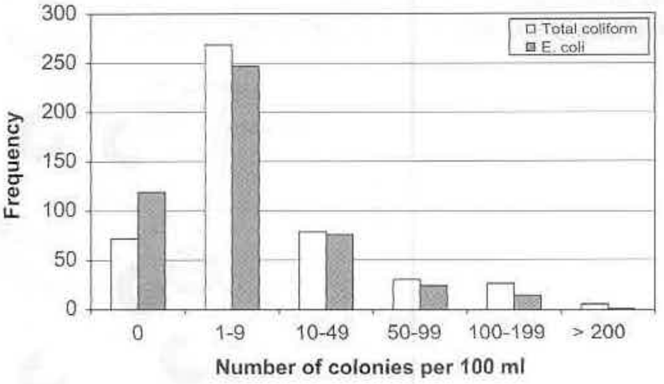


Fig. 2. Summary of bacterial enumeration data for total coliform and *E. coli*.

Table 1
The mean, median and range of $\log_{10}$ total coliform and $\log_{10}$ *E. coli* in stream water collected from the major sampling sites on (i) Derry Burn, (ii) Luibeg Burn and (iii) Lui Water

	Derry Burn (sites 1–5)	Luibeg Burn (sites 6 and 7)	Lui Water (sites 8–12)
Number of samples	139	58	178
(i) *Total Coliform*			
Mean	0.770	0.622	0.761
Median	0.602	0.477	0.699
Range	0–2.79	0–1.90	0–2.40
% detection	75%	81%	78%
(ii) *E. Coli*			
Mean	0.635	0.457	0.617
Median	0.477	0.301	0.477
Range	0–2.32	0–1.60	0–2.26
% detection	63%	59%	65%

5 km downstream. Compared to Derry Burn, there is much less wild camping along the banks of Lui Water and the River Dee. However, there is a mountain hut on the banks of Lui Water (Fig. 1) and wild camping occurs close to site 12 where the public road crosses the Lui Water. The area around site 12 is a very popular picnic location as access to the river, which is shallow (compared to the River Dee) is very easy and it is not far from the car park (Fig. 1). The mean, median and range of total coliform and *E. coli* in stream water samples collected from Derry Burn (sites 1–5), Luibeg Burn (sites 6 and 7) and Lui Water (sites 8–12) is presented in Table 1. Concentrations of both total coliform and *E. coli* were smallest and least variable in samples collected from Luibeg Burn. Total coliform and *E. coli* concentrations in samples collected from Derry Burn and Lui Water were very similar.

Fig. 3 provides a schematic of the spatial distribution of the extent of positive samples for total coliform and *E. coli* at the 12 major sampling sites around Derry Lodge. Overall, site 12 which is located just above the confluence of Lui Water with the River Dee, had the highest number of samples that tested positive for total coliform and *E. coli*, closely followed by site 11 which is located directly in front of the mountain hut (Fig. 3). While this study was being carried out, it was observed that in addition to wild camping at Derry Lodge, many people were camping on the roadside.

3.3. Temporal variations in the microbial status of natural waters

The monthly mean, median and range of total coliform and *E. coli* concentrations in stream water collected during

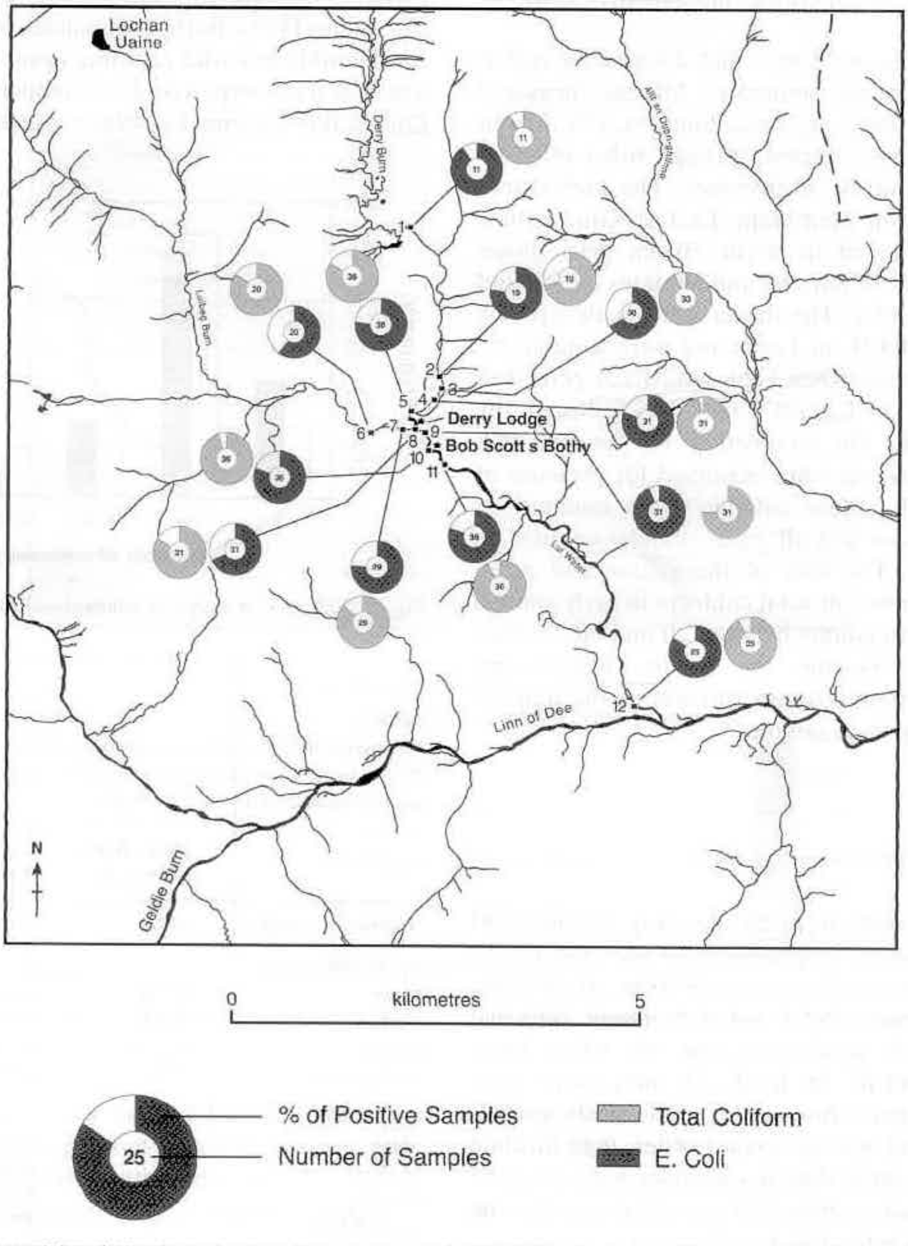


Fig. 3. Map showing the spatial variation in proportion of samples that tested positive for total coliform and *E. coli* at the 12 major sampling sites.

the study are presented in Table 2. Total coliform displayed a wide variation in concentration in all months, with concentrations varying by up to a factor of 100. In general, concentrations were larger in the summer months (June to September) than the winter months (October and March). Although no apparent seasonal trend was observed due to a large number of samples containing low concentrations of total coliform in August, the data clearly show that the maximum value of total coliform display a seasonal pattern, with highest values observed in summer (July, August and September). The monthly range and variability in *E. coli* concentration were very similar to total coliform (Table 2).

The daily mean, median and range of total coliform and *E. coli* concentrations in stream water collected during the study are presented in Table 3. Concentrations of both total coliform and *E. coli* were larger and more variable in stream waters collected on Saturday and Sunday compared to the other days of the week. Both total coliform and *E. coli* were significantly different ($P<0.001$) between the weekend (Saturday and Sunday) and the weekdays (Monday through Friday). This significant difference is also observed between the long-weekend (Saturday and Sunday including Friday and/or Monday) and the remainder of the weekdays.

4. Discussion

As the majority of surface water sampling programmes only analyse for bacteria in urban and agricultural areas (Hunter and McDonald, 1991a; Hunter et al., 1999), which are highly developed and easily accessible, there is a lack of information on total coliform and *E. coli* concentrations in remote upland and wilderness areas. Therefore, the results from this study provide unique information on the range, magnitude and temporal and spatial distribution of total coliform and *E. coli* concentrations in a UK wilderness area. The sites examined in this research are amongst the most remote and 'pristine' (in terms of active management) locations in the UK. Even in such sites, however, the majority of river samples tested positive for the presence of total coliform (85%) and *E. coli* (75%).

Results from other studies that have determined total coliform and *E. coli* in natural waters from wilderness areas are presented in Table 4 for comparison. The number of samples that tested positive for the presence of total coliform and *E. coli* in this study was considerably higher than that reported for a similar study in Grand Teton National Park in the USA (Tippets, 2000, Unpublished Report), where 65% of the 218 samples collected from 26 sites tested positive for faecal coliform and the maximum

Table 2
The monthly mean, median and range of (i) $\log_{10}$ total coliform and (ii) $\log_{10}$ *E. coli* in all stream water samples

	March	June	July	August	September	October
Number of samples	53	40	59	135	108	86
(i) *Total coliform*						
Mean	0.412	0.829	1.092	0.651	0.877	0.382
Median	0.301	0.954	1.041	0.477	0.778	0.301
Range	0–1.322	0–1.560	0–2.146	0–2.419	0–2.790	0–1.785
% detection	75.5	95.0	98.3	90.4	85.2	72.1
(ii) *E. coli*						
Mean	0.407	0.795	0.878	0.499	0.668	0.289
Median	0.301	0.845	0.699	0.301	0.602	0.000
Range	0–1.301	0–1.431	0–2.146	0–2.322	0–2.176	0–1.699
% detection	75.5	92.5	93.2	80.7	68.5	59.3

Table 3
The daily mean, median and range of (i) $\log_{10}$ total coliform and (ii) $\log_{10}$ *E. coli* in all stream water samples

	Mon	Tue	Wed	Thur	Fri	Sat	Sun
Number of samples	83	59	30	30	78	107	94
(i) *Total coliform*							
Mean	0.601	0.416	0.568	0.322	0.579	0.975	0.897
Median	0.477	0.301	0.477	0.301	0.602	0.778	0.699
Range	0–2.491	0–2.000	0–1.672	0–1.255	0–1.732	0–2.790	0–2.362
% detection	78.3	88.1	86.7	66.7	85.9	88.8	92.6
(ii) *E. Coli*							
Mean	0.488	0.285	0.405	0.231	0.422	0.818	0.773
Median	0.301	0.000	0.000	0.000	0.301	0.778	0.477
Range	0–2.322	0–1.778	0–1.568	0–1.146	0–1.672	0–2.176	0–2.146
% detection	68.7	74.6	63.3	50.0	80.8	79.4	88.3

Table 4
The bacterial status of natural waters from other wilderness areas

Study area	Number of samples	TC positive (%)	EC positive (%)	Proportion of samples with > or < 100 TC/EC per 100 ml	Reference
Cairngorms National Park, UK (this study)	482	85	75	> 100 TC/100 ml = 6.4% > 100 EC/100 ml = 3.1%	
Great Smokey Mountains, USA	367		98.6	> 100 ED or FC/100 ml = 80%	Silsbee and Larson (1982)
Logan River, North Utah, USA				< 100 TC/100 ml = 100%	Colthorp and Darling (1975)
Stones River, Tennessee, USA	100	87	62		Brown and Broughton (1981)
Avery Park, Monteith Park, Oregon			11, 21 42, 79		van Ess and Harding (1997)
Grand Teton National Park, USA	218		65		Tippets (2000)

TC = Total coliform, EC = *E. coli*.

value of faecal coliform colonies exceeded 50 at only four of the sites. In this study, maximum total coliform concentrations exceeded 100 cfu/100 ml at 11 of the 59 sites and *E. coli* concentrations exceeded 100 cfu/100 ml at six sites. In the streams of the Great Smoky Mountains National Park 80% of the samples had bacterial densities greater than 100 cfu/100 ml (Silsbee and Larson, 1982), compared to only 6.4% of samples in this study had densities greater than 100 total coliform per100 ml and 3.1% of samples had densities greater than 100 *E. coli* per100 ml (Table 4). This large difference in bacterial concentration may reflect the greater number of visitors to the Great Smoky Mountains National Park compared to Mar Lodge (Table 4). The data in Table 4 also highlights that more samples were collected and analysed for bacteria in this study compared to other studies of bacterial concentrations in wilderness areas.

It is a common perception that water from streams in wilderness areas is bacterially clean and safe to drink, particularly if it is upstream of any wild camping areas and/or areas heavily used by the public. However, the results of this research indicate that this is not the case. For example, even at site 7 which is upstream of the main wild camping area at Derry Lodge, *E. coli* was detected in 9 of the 11 samples (Fig. 3). Thus in wilderness areas, even upstream sources reflect wildlife and deer faecal inputs and so are not always safe for drinking water. The sampling sites around the major camping area at Derry Lodge (sites 8–12) appear likely to be the most risky place from which water for consumption was taken. Visitors potentially contaminate nearby waters by using the ground within and adjacent to this major campsite for the disposal of human excrement. Of the 175 samples taken from sites 8 to 12, *E. coli* was detected in 127. Bob Scott's Bothy has a toilet which is believed to be connected to a septic tank. Of the 83 samples taken from sampling sites close to the Bothy, the presence of *E. coli* was detected in 71. This may be due to upstream contamination from the campground or it may also be contaminated by leachates/spills/overflows from the septic tank. Three sites, adjacent to a popular footpath that runs along side Luibeg Burn, were not used for overnight stays, being neither campgrounds nor bothies, but, for families with children, are popular sites to play and have picnics and were sampled 31, 25, and 33 times. *E. coli* was detected in 29, 19, and 28 of the sample respectively (Fig. 3).

In this study, bacterial concentrations displayed strong temporal trends on a monthly and daily timescale. Concentrations were significantly higher in summer than winter despite an expectation that deer grazing the river flats in winter when visitors were absent might have caused a winter peak. It is possible, in this little researched environment, that lower flows in summer may lead to higher bacterial concentrations but the normal outcome of low summer flows is low bacterial concentrations since the bacterial concentration increase is, in most studies, very sensitive to increased flow. Daily values showed that concentrations were significantly higher at the weekends compared to weekdays. Many other studies, both in wilderness and agricultural catchments have reported seasonal fluctuations in bacterial concentrations with higher numbers being observed during the summer (and autumn) months in well-waters, beach waters, streams, springs and lakes than in winter months (Rutter et al., 2000; Sheehan and Badcock, 1993; Silsbee and Larson, 1982; Skinner et al., 1974). In the Yorkshire Dales, Hunter and McDonald (1991b) also observed that faecal coliform concentrations in stream water displayed a strong seasonal trend, with highest concentrations in summer (mid June to the end of August) and lowest concentrations in winter and spring (mid November to mid June).

Coliform populations in streams have been reported to fluctuate on a daily basis as well as seasonally (Cilimburg et al., 2000). These short-term variations in coliform counts are usually associated with either storm events or periods of heavy recreational use. However, Christensen et al. (1979) reported that higher bacterial levels were observed in stream waters in Greenwater watershed, Washington at the weekends when more campers were present. The results from this study show a similar relationship, with significantly higher concentrations observed at the weekend (with and without Friday and/or Monday) than on weekdays. Although no daily visitor numbers were available for Mar Lodge Estate, observations of tent numbers during the study show that more tents were generally present at the weekend than during the week (Fukasawa, 2004).

5. Conclusions

The sites investigated here are amongst the most remote and unspoilt that can be found in the UK but even in these locations indicator bacteria are found frequently (75% positive with some values in excess of 200 *E. coli* per 100 ml). While this contamination appears to be sporadic in streams on the high tops, at lower elevation but still above any agriculture usage, the presence of indicator bacteria in water samples are near ubiquitous. There appears to be evidence that the contamination is associated with wild camping, both through the higher values in proximity to these sites and through the seasonal and weekly concentration towards those periods of occupancy. This is exacerbated by the policy of having no disposal facilities on site in order to preserve wilderness characteristics. The policy of the 'long walk in' to preserve such remote sites in Scotland means that walkers are unlikely to be able to carry sufficient potable water and resort to the use of the contaminated streams. Advice on best practice for water use and waste disposal is clearly required. Given that this site is one of the cleanest and most pristine locations in the UK, it is highly likely that this is an issue to be encountered at any site where wild camping takes place. It exposes a 'tension' that will be difficult to manage namely a choice between (i) banning overnight stays on such sites (but this is in effect a ban on access) or (ii) the provision of water and toilet facilities which will have the effect of significantly reducing the wilderness value of the site.

Acknowledgements

We are grateful to the staff of CREH *Analytical* Limited and in particular to John Watkins and Carol Frances for the provision of specialist advice, training, equipment and consumables. The work could not have been completed without the cooperation and advice of the staff rangers and manager of the National Trust for Scotland at Mar Lodge. Diagrams were produced by the graphics unit at the School of Geography, Leeds University. The encouragement and advice of the editor and the reviewers was gratefully received.

References

Brown, L.C., Broughton, C.W., 1981. A survey of pollution indicator bacteria in water wells of Rutherford Country, Tennessee. Journal of the Tennessee Academy of Science 56 (3), 73–75.

Christensen, H.H., Pacha, R.E., Varness, K.J., Lapen, R.F., 1979. Human use in a dispersed recreation area and its effect on water. In: Recreational Impact on Wildlands Conference Proceedings. Forest Service, Pacific Northwest Region, Portland, Oregon, USA, pp. 107–119.

Cilimburg, A., Monz, C., Kehoe, S., 2000. Wildland recreation and human waste: a review of problems practices and concerns. Environmental Management 25 (6), 587–598.

Cole, D.N., 1983. Assessing and Monitoring Backcountry Trail Conditions. USDA Forest Service Research Paper, INT-303:1-10. USDA Forest Service, Washington, DC.

Cole, D.N., 1990. Ecological impacts of wilderness recreation and their management. In: Hendee, J., Stankey, G., Lucas, R. (Eds.), Wilderness Management, pp. 425–466.

Cole, D.N., 1991. Changes on Trails on the Selwey-Bitterroot Wilderness, Montana, 1978–1989. USDA Forest Service Research Paper INT-450:1–5.

Cole, D.N., 1996. Ecological manipulation in the wilderness: an emerging management dilemma. International Journal of Wilderness 2 (1), 15–18.

Colthorp, G.B., Darling, L.A., 1975. Livestock grazing–a nonpoint source of water pollution in rural areas. In: Water Pollution Control in Low Density Areas: Proceedings of a Rural Environmental Engineering Conference, Vermont. University Press of New England, Hanover, New Hampshire, USA, pp. 341–358.

CREH, 1999. A Procedure for the Microbiological Analysis of Water for Coliforms, Faecal Coliform and *Escherichia coli*. CREH Leeds, UK, p. 30.

Curry-Lindahl, 1990. The Cairngorms National Nature Reserve: the Foremost British Conservation Area of International Importance. In: Conroy, J., Watson, A., Gunson, A. (Eds.), Caring for the High Mountains. Centre for Scottish Studies, University of Aberdeen, Aberdeen, UK, pp. 61–90.

Fukasawa, K., 2004. The management of bacterial issues in wilderness waters: A case study in the forest of Mar, Scotland. Unpublished Ph.D thesis, University of Leeds.

Gary, H.L., Adams, J.C., 1985. Indicator bacteria in water and stream sediments near the snowy range in southern Wyoming. Water, Air and Soil Pollution 25, 133–144.

Hammit, W.E., Cole, D.N., 1998. Wildland Recreation: Ecology and Management, second ed. Wiley, New York, p. 361.

Hampton, B., Cole, D.N., 1995. 'Softpaths: How to Enjoy the Wilderness without Harming it'. Stackpole Books, Harrisburg, PA, p. 173.

Hendee, J., Stankey, G., Lucas, R., 1990. Wilderness Management. North American Press, Colorado, USA, p. 546.

Hermann, R., Williams, O.R., 1986. Water resources research for wilderness: a state of knowledge review. In: Hermann, R., Williams, O.R. (Eds.), Proceedings of the National Wilderness Research Conference: Issues, State-of-Knowledge, Future Directions. USDA Forest Service Intermountain Research Station General Technical Report INT-220. 191–202pp.

Hunter, C., McDonald, A., 1991a. The occurrence of coliform bacteria in the surface soils of two catchment areas in the Yorkshire Dales. Journal of Water and Environmental Management 5 (5), 534–538.

Hunter, C., McDonald, A., 1991b. Season changes in the sanitary bacterial quality of water draining a small upland catchment in the Yorkshire Dales. Water Research 25 (4), 447–453.

Hunter, C., Perkins, J., Tranter, J., Gunn, J., 1999. Agricultural land use effects on the indicator bacterial quality of an upland stream in the Derbyshire Peak District in the UK. Water Research 33 (17), 3577–3586.

IUCN, 1994. Guidelines for Protected Area Management Categories. International Union for the Conservation of Nature, Glad, Switzerland and Cambridge, UK.

Kuss, F.R., Graefe, A.R., Vaske, J.J., 1990. Visitor Impact Management: A Review of Research, vol. 1. National Parks and Conservation Association, Washington, DC, 256p.

McDowell, T.R., 1979. Geographic Variations in Water Quality and Recreation Use Along the Upper Wallow River and Selected Tributaries. PhD Thesis. Oregon State University.

Rutter, M., Nicholas, G.L., Swan, A., De Louvois, J., 2000. A survey of the microbiological quality of private water supplies in England. Epidemiology and Infection 124 (3), 417–425.

Sartory, D., Watkins, J., 1999. Conventional culture for water quality assessment: is there a future? Journal of Applied Microbiology 85, 225s–233s.

Sheehan, D., Badcock, C., 1993. The Influence of Some Environmental Factors on Densities of Faecal Coliforms at Fourteen Sydney Beaches. Report No. 93/28. AWT Science and Environment, Water Board, Sydney, Australia.

Silsbee, D.G., Larson, G.L., 1982. Bacterial water quality: springs and streams in the Great Smoky Mountains National Park. Environmental Management 6 (4), 353–359.

Skinner, Q.D., Adams, J.C., Rechard, P.A., Beetle, A.A., 1974. Effect of summer use of a mountain watershed on bacterial water quality. Journal of Environmental Quality 3, 329–335.

Tippets, N., 2000. Backcountry water quality testing in Grand Teton National Park: 1999 summer season. Unpublished Report.

van Ess, E., Harding, A.K., 1997. A bacteriological study of natural freshwater swimming areas in western Oregon. Indian Journal of Environmental Health 39 (3), 14–19.

Varness, K.J., Pacha, R.E., Lapen, R.F., 1978. Effects of dispersed recreational activities on the microbiological quality of forest surface water. Applied and Environmental Microbiology 36 (1), 95–104.

Watanabe, T., Fukasawa, K., 1998. Seven year deterioration of hiking trails and measures to mitigate soil erosion. Mount Kurodake, Daisetsuzan National Park, Hokkaido, northern Japan. Geographical Review of Japan 71A (10), 753–764.

Watkins, J., Jian, X., 1997. Cultural methods of detection for microorganisms: recent advances and successes. In: Sutcliffe, D.W. (Ed.), Microbiological Quality of Water. Freshwater Biological Association, Ambleside, Cumbria.

ELSEVIER

Journal of Environmental Management 87 (2008) 609–622

Journal of
Environmental
Management

www.elsevier.com/locate/jenvman

Prioritisation of farm scale remediation efforts for reducing losses of nutrients and faecal indicator organisms to waterways: A case study of New Zealand dairy farming

R.M. Monaghan, C.A.M. de Klein, R.W. Muirhead*

AgResearch, Invermay Agricultural Centre, Private Bag 50034, Mosgiel, New Zealand

Received 18 January 2006; received in revised form 25 July 2006; accepted 30 July 2006

Abstract

The international competitiveness of the New Zealand (NZ) dairy industry is built on low cost clover-based systems and a favourable temperate climate that enables cows to graze pastures mostly all year round. Whilst this grazed pasture farming system is very efficient at producing milk, it has also been identified as a significant source of nutrients (N and P) and faecal bacteria which have contributed to water quality degradation in some rivers and lakes. In response to these concerns, a tool-box of mitigation measures that farmers can apply on farm to reduce environmental emissions has been developed. Here we report the potential reduction in nutrient losses and costs to farm businesses arising from the implementation of individual best management practices (BMPs) within this tool-box. Modelling analysis was carried out for a range of BMPs targeting pollutant source reduction on case-study dairy farms, located in four contrasting catchments. Due to the contrasting physical resources and management systems present in the four dairy catchments evaluated, the effectiveness and costs of BMPs varied. Farm managements that optimised soil Olsen P levels or used nitrification inhibitors were observed to result in win–win outcomes whereby nutrient losses were consistently reduced and farm profitability was increased in three of the four case study farming systems. Other BMPs generally reduced nutrient and faecal bacteria losses but at a small cost to the farm business. Our analysis indicates that there are a range of technological measures that can deliver substantial reductions in nutrient losses to waterways from dairy farms, whilst not increasing or even reducing other environmental impacts (e.g. greenhouse gas emissions and energy use). Their implementation will first require clearly defined environmental goals for the catchment/water body that is to be protected. Secondly, given that the major sources of water pollutants often differed between catchments, it is important that BMPs are matched to the physical resources and management systems of the existing farm businesses.

Keywords: Dairy; Grazing; N mitigations; P; Faecal bacteria; Water quality; Farm best management practices

1. Introduction

Agricultural non-point source pollution of waterways is recognised as a major cause of water quality degradation in many NZ regions (Vant, 2001; Hamill and McBride, 2003). Intensive agriculture is known to emit significant amounts of nutrients, particularly nitrogen (N) (Ledgard et al., 1999; Monaghan et al., 2005b; Scholefield et al., 1993) and phosphorus (P) (Gillingham and Thorrold, 2000; Monaghan and Smith, 2004; Sharpley and Syers, 1979), faecal bacteria (Monaghan and Smith, 2004) and sediment. Whilst these emissions are typically not large by agronomic standards, the transfer of these pollutants from land to water can result in significant water quality impairment (Larned et al., 2004; Wilcock et al. 2006). In the past decade much research has focussed on the role of dairy farming as a contributor to nutrient enrichment of ground and surface waters (Wilcock et al., 1999; Hamill and McBride, 2003; de Klein, 2005). Whilst dairy cows are never the sole contributor to water quality impairment, this research has shown that inappropriate management of the dairy farm system has the potential to cause significant

---

*Corresponding author. Tel.: +64 3 4899261; fax: +64 3 4893809.
E-mail address: richard.muirhead@agresearch.co.nz (R.W. Muirhead).

0301-4797/$ - see front matter © 2007 Elsevier Ltd. All rights reserved.
doi:10.1016/j.jenvman.2006.07.017

groundwater and stream pollution (Wilcock et al., 2006). In response to market signals the dairy farming sector has in recent years grown in size and intensity, with land area, animal stocking rate and per hectare production increasing by 32, 11 and 32% between 1992 and 2004, respectively (LIC, 2005). The combined effect of these increases has inevitably led to greater leakage of nutrients from the dairy farm system (Ledgard et al., 1999). With the resulting increase in pressure on the water resource, mitigation practices that reduce farm losses of nutrient and pathogenic micro-organisms are urgently sought. Community concerns about the current trajectory of farming and the resulting pressure on the environment are encapsulated in a recent report from the NZ Parliamentary Commissioner for the Environment (PCE, 2004), which describes the environmental and economic risks associated with a continued increase in non-point source pollution from farming. The wider public are also cognisant of the requirement that farm mitigation measures allow farm profitability to be maintained for this important industry.

As a first formal step to improve the environmental performance of most of New Zealand's 13,000 dairy farms, the dairy industry has developed the Clean Streams Accord which is an agreement between the industry, local regulatory agencies and central government authorities to implement a number of key environmental goals (Fonterra et al., 2003). These initial goals have a particular focus on measures to protect aquatic and riparian habitat and to have systems in place that can balance farm nutrient inputs and outputs. The next challenging step to improved farm environmental performance is to develop a farm nutrient management system that can identify the most cost–effective way for farms to continue to lift productivity (and thus profitability) without the consequent increase in farm leakage of nutrients and pathogenic micro-organisms. The purpose of this paper is to outline some of the environmental mitigation measures, further to the Clean Streams Accord, that have been identified as relevant to the dairy farming sector, and to present a cost–benefit analysis of these options, including a consideration of the logistical implications that arise from the introduction of such measures. The paper presents an analysis of farm management options to mitigate sources of nutrient and faecal indicator organisms that are transferred to waterways. Using four case study dairy farms and catchments, prioritisation of these mitigation measures is discussed within the context of current dairy farm management practices and the effects-based policy context that determines where and how remedial actions are made. The effect of the mitigation measures on other environmental concerns such as greenhouse gas emissions and energy use is also discussed as a consideration for prioritising the measures.

2. Case study farms and catchments

In 2001, the Best Practice Dairying Catchments project was established to integrate environmentally sustainable practices into dairy farming. This project is carried out in four dairy catchments in New Zealand, two in the North Island and two in the South Island (Fig. 1), to study farm productivity and catchment-specific environmental issues. The focus of the study is on water quality issues, although estimates of greenhouse gas (N_2O, CH_4 and CO_2) emissions are also made to assess the wider environmental impact of dairy farming in these catchments. The whole-farm system approach of this project enables an evaluation of dairy systems that optimise farm productivity, whilst minimising environmental impacts. For each catchment, detailed information of farm management practices, animal production, fertilizer usage and soil management was obtained through farm surveys on 7–20 dairy farms in each catchment. Pasture growth and pasture quality monitoring information was obtained from two monitor farms within each catchment for three successive years, and surveys of soil chemical, biological and physical properties have been conducted biannually since 2001.

The farm and soil survey information together with pasture monitoring data were used to calculate average farm size, stocking rate, production and management practices for each catchment. The data of the average farm was then used with various modelling tools to describe the economic and environmental performance of the 'average' case study dairy farm within each catchment (Fig. 2). Firstly, the UDDER dairy farm simulation model (Hart et al., 1998; Larcombe, 1999) was used to characterize farm production by simulating herd characteristics, pasture growth and feed intake, milk production and changes in cow body condition score. The pasture and milk production outputs from UDDER and the soil and farm

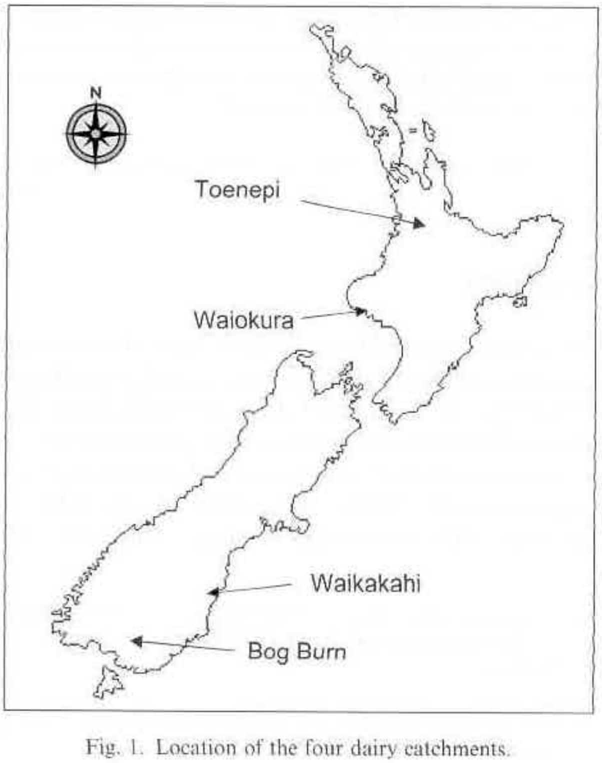


Fig. 1. Location of the four dairy catchments.

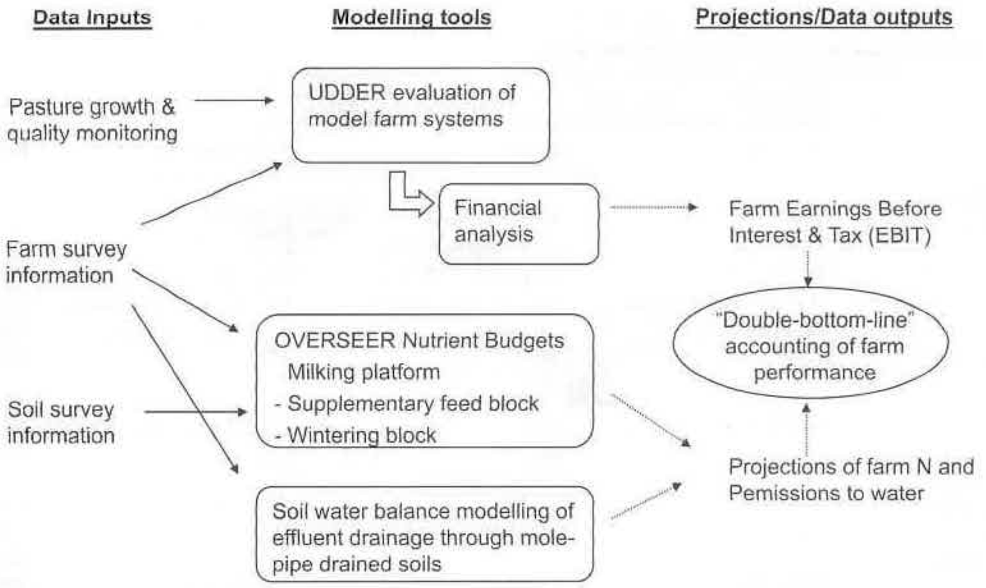


Fig. 2. Schematic representation of modelling and assessment process.

information from the surveys were then used within the OVERSEER® nutrient budget model (McDowell et al., 2005; Wheeler et al., 2003) to calculate annual budgets of nutrients, greenhouse gas emissions and energy use. Finally, a purpose-built farm financial model was used to calculate a full farm financial budget based on the current value of produce (milk and meat) and production expenses (e.g. imported feed, off-farm grazing) (Dexcel, 2003; MAF, 2003). Farm profit was expressed as Earnings Before Interest and Tax (EBIT). The dairy "systems" evaluated here included the home farm on which the cows were milked and off-farm areas used for supplement production and/or wintering of cows. Estimates of P and N emissions from the latter, for which the OVERSEER® model is not currently set up to accurately represent, were derived from on-going field trials and values reported in the literature (McDowell et al., 2003a, b, 2005; R Monaghan, unpublished results). Energy flows (including the energy embodied in the manufacture of fertilizer N) and greenhouse gas emissions were also estimated for each part of the dairy system, again using the OVERSEER® model or values derived from the literature for areas which were not encompassed by OVERSEER® (de Klein et al., 2002; Wells, 2001). The "average" farm described for each catchment was then used as a reference or control farm for assessing the effectiveness of implementing a range of BMPs that targeted the reduction of N, P and/or faecal bacteria losses.

Characteristics of each catchment and case study farm are presented in Table 1. The North Island catchments (Toenepi and Waiokura) are warmer and wetter, and animals are generally pasture-grazed throughout the winter within the dairy farm. In the cooler South Island catchments (Waikakahi and Bog Burn), the cows are generally grazed off-farm on forage crops during winter. In the Waikakahi catchment, annual rainfall is low and many farms are irrigated, most commonly using flood-irrigation systems (border dyke). In all catchments, supplementary feed is imported onto the farm, with the Waikakahi case-study farm importing substantially more feed than the other farms. The North Island case-study farms use a combination of a two-pond effluent treatment system and land application for managing the farm dairy effluent (FDE) collected at the milking parlour, while the South Island farms only use land application. Estimated nitrate leaching and P losses from the case-study farms range between 25 and 48 kg N/ha/year, and between 0.4 and 1.4 kg P/ha/year. Any N and P losses from areas used for off-farm wintering or for the production of supplements are included in this assessment of the total environmental (and economic) performance of each case-study farm (Table 1).

3. Potential mitigations: the BMP toolbox

The most effective mitigation strategies are those that address the main sources of contaminants within a system. Fig. 3 provides an overview of the main sources of contaminants and their potential mitigation options in New Zealand dairy systems.

3.1. P and FDE management

New Zealand dairy systems are characterised by clover-based pasture and year-round grazing with minimal housing of animals. As a result, P fertiliser usage is relatively high to maintain adequate soil P fertility for optimum clover growth. Direct losses from P fertiliser application are generally low because best practices for fertiliser management are already fine-tuned (NZFMRA, 2002). An exception is in systems that have large volumes of overland flow, such as irrigation-wash exiting border

Table 1
Characteristics of the four dairy catchments and their case study farms

	Catchment			
	Toenepi	Waiokura	Waikakahi	Bog Burn
Catchment characteristics				
Mean annual temp (°C)	13.3	12.7	10.7	10.2
Catchment area (km^2)	15.5	23.5	41.0	24.7
Topography	Rolling	Flat	Flat	Flat
Major soil type	Well drained volcanic silt loam	Well drained volcanic silt loam	Well drained sandy and silt loams (sedimentary)	Poorly drained silt loam (sedimentary)
Average annual rainfall (mm)	1200	1400	540	921
Farm characteristics				
Irrigation (mm/year)	Nil	Nil	810	Nil
Fertilizer N use (kg/ha/year)	73	88	172	72
Clover N fixation (kg/ha/year)	142	136	32	130
Fertilizer P use (kg/ha/year)	61	65	60	65
On-farm pasture area (ha)	70	75	213	218
Off-farm forage crop area (ha)	0	0	45	44
Animal winter grazing[a]	On farm	On farm	70% off-farm	100% off-farm
Cow milking days per year	265	275	283	269
Stocking rate (cows/ha)	2.9	3.4	3	2.9
Milk production (L/ha)	10,645	11,595	12,890	12,230
Milk production (L/cow)	3655	3350	4170	4220
Imported feed (t DM/ha)	0.9[b]	0.3[c]	2.5[b]	0.2[d]
FDE[e] management system	Pond treatment or land application	Pond treatment or land application	Land application	Land application
Soil Olsen P (mg/L)	53	65	46	42
Key Farm environmental performance indicators[f]				
N leaching				
kg/ha	32	48	52	30
kg/T milksolids[g]	37	49	65	34
P loss				
kg/ha	1.41	0.39	1.25	0.94
kg/T milksolids	1.66	0.41	1.57	1.08

[a] Off-farm: animals grazed off-farm on forage crops.

[b] Mainly pasture silage @ NZ\$0.17/kg DM (NZ\$1 ≈ US\$0.68).

[c] Maize or cereal silage @ NZ\$0.25/kg DM.

[d] Grain @ NZ\$0.30/kg DM.

[e] FDE: farm dairy effluent.

[f] Includes estimates of losses from off-farm areas for winter grazing and supplement production.

[g] One kilogram of milksolids derived from 12 l of milk.

dyke-irrigated pastures, where P losses from soluble P fertilisers can be potentially substantial (McDowell et al., 2003b). In these situations, the use of low solubility P fertilisers (McDowell and Catto, 2005) and improved bunding (dykes around the perimeter of a paddock to prevent excess irrigation water leaving the paddock) of the border dyke land could reduce P losses in overland flow from grazed pastures (Table 2). In other situations soil and sediment losses from high P fertility soils can make a substantial contribution to P loss (McDowell et al., 2003a; Morton et al., 2003). Maintaining soil P fertility at recommended levels could therefore reduce the risk of P losses from dairy pastures (Table 2), particularly for farms in the Toenepi and Waiokura catchments where soil Olsen P values are well above economic optima (Table 1). Economic optima are defined here as the soil Olsen P levels where farm EBITs are maximised, and vary as a function of farm production and soil type (Edmeades et al., 2006). The high soil P levels found on some of the catchment farms are a reflection of excessive historic maintenance P fertiliser inputs.

Other important sources of P are farm dairy effluent (FDE) treatment ponds, FDE applied to land, and dung patches deposited during grazing. With the recognition that FDE is a valuable source of nutrients, best management practices were developed for applying FDE to land. These included recommendations on maximum annual rates of N application (and thus the farm area required to receive effluent loadings), split applications and exclusion times for animals after application (e.g. Cameron et al., 1999; Longhurst et al., 2000; Roach et al., 2001). Further research suggested that when following these recommendations, effluent application to land is relatively effective with only 2–20% of the nutrients applied within

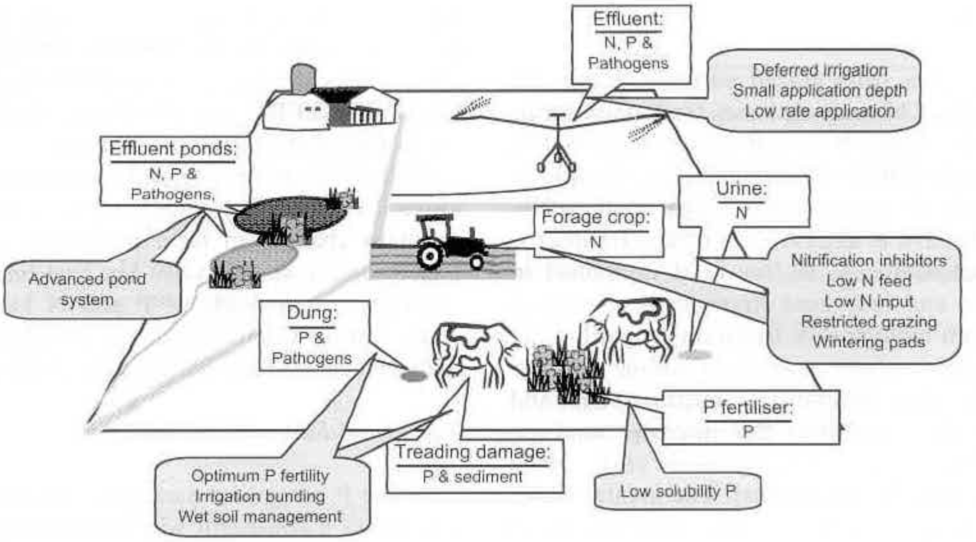


Fig. 3. Schematic representation of the main sources of contaminants (white boxes) and potential mitigation strategies (grey boxes) on dairy farms in New Zealand (de Klein, 2005).

Table 2
Potential dairy farm source mitigation measures

Measure	System target	Pollutant reduction	Reference
1. Optimum soil Olsen P level	Grazed pasture	P	McDowell et al. (2003a); Morton et al. (2003)
2. Deferred effluent irrigation	Land application of FDE to wet soil	Faecal bacteria, suspended solids, ammonium-N, P	Houlbrooke et al. (2004a); Monaghan and Smith (2004).
3. Applying small amounts/depths of effluent	Land application of FDE to wet soil	Faecal bacteria, suspended solids, ammonium-N, P	Houlbrooke et al. (2004a); Monaghan and Smith (2004).
4. Low rate effluent irrigation	Land application of FDE to wet soil, or soils with low infiltration rates.	Faecal bacteria, suspended solids, ammonium-N, P	Monaghan et al. (2005a)
5. Bunding of border dyke irrigated land	Border dyke-irrigated grazed pasture	P, faecal bacteria	Carey et al. (2004).
6. Low solubility P fertiliser	Border dyke-irrigated grazed pasture	P	McDowell et al. (2003b); Carey et al. (2004).
7. Nitrification inhibitors	Grazed pasture	Nitrate-N	Di and Cameron (2003, 2005).
8. Inclusion of low N feed in diet	Grazed pasture	Nitrate-N	Ledgard et al. (2003, 2004)
9. Low N input dairy farming	Grazed pasture	Nitrate-N	Monaghan et al. (2004).
10. Restricted autumn or winter grazing	Grazed pasture	Nitrate-N	de Klein et al. (2006); Ledgard and Menneer (2005).
11. Covered wintering pads	Grazed forage crop wintering areas	Nitrate-N	Monaghan et al. (2005a).
12. Advanced Pond System (APS)	FDE treatment	Faecal bacteria, suspended solids, ammonium-N, P	Craggs et al. (2004)

FDE being lost to water bodies (work reviewed by Houlbrooke et al., 2004b). However, it was also recognised that heavier-textured soils and soils with mole-pipe drainage systems have an increased risk of preferential flow resulting in higher risk of contaminant losses (Monaghan and Smith, 2004). Discharges of FDE to waterways, either via treatment ponds or direct drainage through mole-pipe drains, can also contribute significant quantities of faecal bacteria and ammonium-N (Hickey et al., 1989; Monaghan and Smith, 2004). Although ammonium-N losses generally represent 5% or less of total farm N losses to waterways (Monaghan et al., 2005b), in some situations the pulsed outputs of ammonium-N in FDE arising from effluent flow through mole-pipe drainage systems can potentially lead to ammonia toxicity to aquatic life. More recent research efforts therefore focussed on developing best management practices for land application of effluent to mole-pipe drained soils (Table 2; Houlbrooke et al., 2004a; Monaghan and Smith, 2004), and on developing an Advanced Pond System (APS) in areas where land application of FDE is unsuitable (Craggs et al., 2004).

3.2. N management

Animal urine patches from grazed pastures remain the largest single source of N leaching losses from dairy farms (e.g. de Klein et al., 2006). It is well documented that N leaching losses from animal urine patches in intensively grazed systems can be substantial (Clough et al., 1996; Fraser et al., 1994; Silva et al., 1999). Nitrogen fertiliser is not a major direct source of N loss as it is applied in relatively low rates and only used strategically to supplement N supply from biological N fixation (Ledgard et al., 1999). In many areas, winter grazing of forage crops is common practice, and preliminary measurement and modelling studies have identified this intensive land-use as another important source of N (Thorrold et al., 1998; R.M. Monaghan, unpublished results). The greater losses from the wintering part of the system arise due to (i) relatively large amounts of mineral N remaining in the soil in late autumn following pasture cultivation and forage crop establishment the preceding spring, and (ii) the deposition of much excretal N onto the grazed forage crop during winter when plant uptake is correspondingly low. Best management practices for mitigating N losses to waterways therefore focus on those that reduce the total amount of urine N deposited by reducing N inputs to the farm system via fertiliser and/or feed, and those that avoid the deposition of urine patches during autumn and winter when the risk of N loss is highest (Ledgard and Menneer, 2005). In addition, the use of the nitrification inhibitor dicyandiamde (DCD) has been promoted in recent years to reduce the risk of N leaching losses from urine patches (e.g. Di and Cameron, 2003, 2005).

4. Effectiveness of mitigation measures

The impact of potential mitigation strategies on the environmental and economic performance of the average case-study farm was assessed by re-running the models using appropriate assumptions (Table 3). For simple engineering-type mitigation measures such as the introduction of a deferred effluent irrigation management strategy, only the OVERSEER® nutrient budget and farm financial models were required to be re-run under the appropriate new assumptions. In contrast, mitigation measures that incurred changes in farm stocking rate and/or feed input required the sequential re-running of all three modelling tools to account for whole-system changes such as pasture surpluses/deficits, feed quality, and additional supplementary feed required. All mitigation measures were costed and expressed on an annualised basis to include the opportunity cost of any additional capital required (8%), depreciation (spread across the life of any new infrastructure required), running and maintenance costs, and additional labour required. For mitigations that required the use of covered feeding and confinement areas (the wintering pad and restricted autumn/winter grazing systems), it was assumed that 85% of feed nutrients deposited on the feedpad were captured and returned to pastures in late spring. Nutrient returns in these manure applications to farm effluent blocks were estimated using the OVERSEER® nutrient budget model and accounted for in farm fertilization programmes.

A summary of the projected reductions in farm P and N losses to waterways under a range of BMPs, and the resultant changes in farm profitability, are presented in Figs. 4 and 5, respectively. The first feature of note is the contrasting magnitude of P and N losses from the case study farms, which reflects the different soil, climate and management systems within each of the catchments.

4.1. P and FDE management

Of the P mitigation measures considered in Fig. 4, only one shows a consistent win–win outcome when evaluated from both economic and environmental perspectives. Reducing maintenance P fertiliser inputs and soil Olsen P levels to economically-optimal levels is projected to result in annual savings of between \$15 and \$22 per cow over a following 10-year period, and annual savings of between \$10 and \$18 per cow thereafter. In the case of the Toenepi and Waiokura catchments where soil Olsen P values are well above economically optimal levels (Tables 1 and 3), this measure is predicted to reduce farm P losses by 30 and 37%, respectively. In the case of the South Island catchments where soil Olsen P levels are closer to economic optima, this measure is projected to reduce P losses from Waikakahi farms by 14% and Bog Burn farms by 7%. Mitigation measures targeting FDE management (mitigations 2, 3, 4, and 12 in Table 2), which is a particularly important source of P emitted from dairy farms in the Toenepi and Bog Burn catchments, incur small reductions in farm EBIT values whilst delivering reductions in farm P losses of between 10 and 55%. We should note here that the performance of an Advanced Pond System (APS) for the treatment of FDE is included in our evaluation of BMPs for completeness. In practice, this technology primarily targets the reduction of losses of faecal bacteria from dairy farms, although for farms that still use two-pond treatment systems this technology can also deliver significant reductions in farm discharges of P and, to a lesser extent, N.

Discharges of FDE via two-pond treatment systems, still commonly used in the Toenepi catchment, account for much of the modelled P loss from these dairy farms, whereas in the Waikakahi catchment irrigation wash from the border dyke-irrigated land is calculated to contribute approximately 70% of annual P losses from dairy farms. As a further contrast, direct drainage of FDE through mole-pipe drains, overland flow and mole-pipe drainage of soil P from the poorly drained Pukemutu soil in the Bog Burn catchment are estimated to make approximately equal contributions to the predicted P loss of 1.3 kg P/ha/year emitted from dairy farms. Paradoxically, the poorly drained nature of the soils in this catchment contribute to

Table 3
Key assumptions made in mitigation modelling analysis

Measure	Assumptions
P source mitigations[a]	
1. Optimum soil P fertility	• Economically optimum soil Olsen P levels defined for each case study farm using the OVERSEER3® Fertiliser Recommendations model: 25, 31, 38 and 34 μg P/ml for Toenepi, Waiokura, Waikakahi and Bog Burn catchments, respectively.
2. Deferred effluent irrigation	• Based on the provision of pond storage costing $37 per cow, depreciated over a 30 year period. Annual maintenance cost of $2.5 per cow.
3. Applying small amounts/depths of effluent	• Additional labour requirement costed at $1.3 per cow per annum.
4. Low rate effluent irrigation	• Annual cost of upgrading from travelling irrigator applicator to K-line applicator valued at $5.8 per cow; cost of up-grading from two-pond treatment system to land application via K-line valued at $2.4/cow/year (nutrient credit of $9.8 assumed).
5. Irrigation bunding	• Bunding of border dyke run ends costed at $80 per ha.
6. Low solubility P fertiliser	• Maintenance P fertiliser typically applied in December. • Fertilization costs of soluble P fertiliser and low solubility P fertiliser assumed to be similar.
N source mitigations	
7. Nitrification inhibitors	• Four percent increase in pasture production, spread across the growing season • Thirty percent reduction in nitrate leaching and 75% reduction in N_2O emissions from grazed pasture areas only • Inhibitor application costed at $126/ha/year.
8. Inclusion of low N feed in diet	• Extra feed grown from fertiliser substituted with low N feed such as maize silage (Toenepi and Waiokura) or cereal silage (Waikakahi and Bog Burn).
9. Low N input dairy farming	• Fertiliser N inputs to case study farms set to nil.
10. Restricted autumn or winter grazing	• Six hour grazing of pastures from mid May to early July in Toenepi and Waiokura catchments. • Four hour grazing of pastures from March til mid May in Waikakahi and Bog Burn catchments. Additional 2 kg DM/cow/day fed to cows whilst on feedpad following evening milking. It was assumed that these southern farms already had a covered feedpad for use as per the Wintering Pad scenario described below. • Eighty-five percent of feed nutrients deposited on feedpad captured and returned to pastures in late spring.
11. Wintering pads	• Animals kept on a covered pad for 10 (Toenepi and Waiokura farms), 8 (Waikakahi) or 9 (Bog Burn farms) weeks during winter. • Supplementary feed required for dry cows during winter: 8 (Toenepi and Waiokura farms) or 9 (Waikakahi and Bog Burn farms) kg DM/cow/day. • Extra feed imported onto Toenepi and Waiokura farms to increase per-cow production; extra winter feed imported onto Waikakahi and Bog Burn farms substituted feed "purchased" from off-farm forage crop areas. • Eighty-five percent of feed nutrients captured on pad and returned to pastures in late spring. • Pad construction cost of $470 per cow.
Mitigations for reducing losses of faecal micro-organisms[a]	
12. Advanced Pond System	• Per cow construction cost ($94) assumed to be constant in all 4 catchments. Depreciated over a 30 year period.

[a] Mitigations 2–5 also target the reduction of faecal micro-organism losses from farm systems.

the relatively low N losses determined for Bog Burn dairy farms, which is in contrast to the considerably larger farm N leaching losses predicted for the free-draining volcanic ash soils in the Toenepi and Waiokura catchments, or the shallow, free-draining sedimentary soils that are common in the Waikakahi catchment. These contrasting physical resources and management systems thus demonstrate that farm-specific mitigation measures are needed to target specific pollutant sources on-farm–one-size does not necessarily fit all.

4.2. N management

Evaluation of management systems that deliver reduced losses of nitrogen (N) from dairy farms in the catchments shows that the use of the nitrification inhibitor

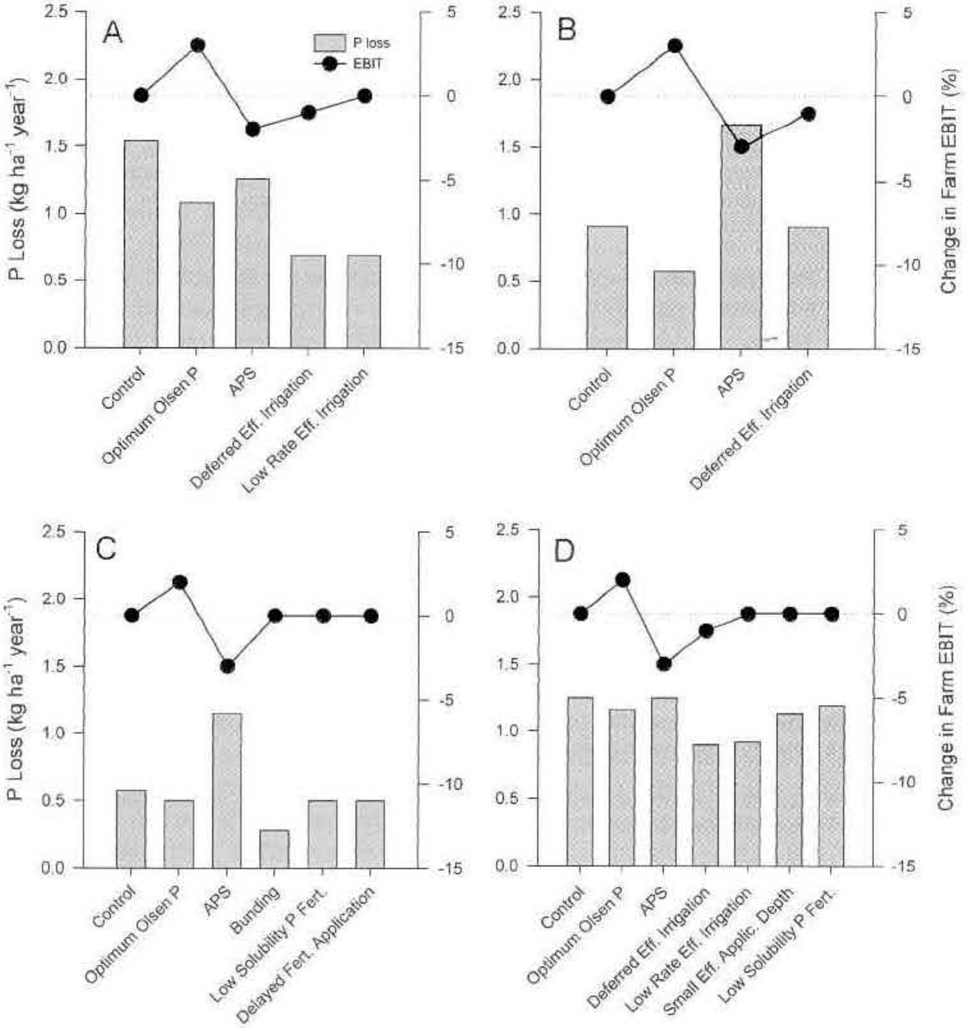


Fig. 4. Total P losses to waterways (bars) and projected cost-effectiveness (symbols) of a range of source management mitigation strategies which target P losses from dairy farms in the (A) Toenepi, (B) Waiokura, (C) Waikakahi and (D) Bog Burn catchments. Dashed line represents nil change in farm earnings before interest and tax (EBIT) values.

DCD is a potentially cost-effective measure in all catchments, increasing farm Earnings Before Interest and Tax (EBIT) by up to 9%. Although the science behind this technology is still at an early stage, their use is projected to reduce farm losses of N in drainage by between 9 and 30%. Wintering pads appear to be a cost-effective management system for farms in the Bog Burn catchment, potentially reducing N losses by more than 30% and increasing farm EBITs by 2%. Given the reductions in EBIT values projected for dairy farms in the other catchments, however, the wider adoption of this management system is currently unlikely. Restricted grazing systems, low N feed supplements and low fertiliser N input management systems are all strategies that are projected to significantly reduce N losses from dairy farms within the catchments. At this stage, however, the economics of these management systems generally incur small reductions in farm profitability.

4.3. Sources and mitigation of faecal pollution

Our current scientific understanding of the sources and pathways of faecal bacteria losses from dairy farms is relatively poor in comparison to our understanding of sources and pathways of nutrient loss (Wilcock et al., 1999). Nonetheless, some obvious sources have been identified that provide a few clues as to where mitigations might be best targeted in the first instance. Documented sources of faecal bacteria transferred from grazed pastoral soils to waterways include the rapid drainage loss of irrigated FDE applied to soils that exhibit a high degree of preferential flow (McLeod et al., 2003, 2004; Monaghan and Smith, 2004), rain-fed mole-pipe drainage (Monaghan and Smith, 2004), border dyke wash (Carey et al., 2004) and effluent pond discharges (Craggs et al., 2004; Hickey et al., 1989). Overland flow is also believed to be a major contributor of the total microbial load delivered to streams

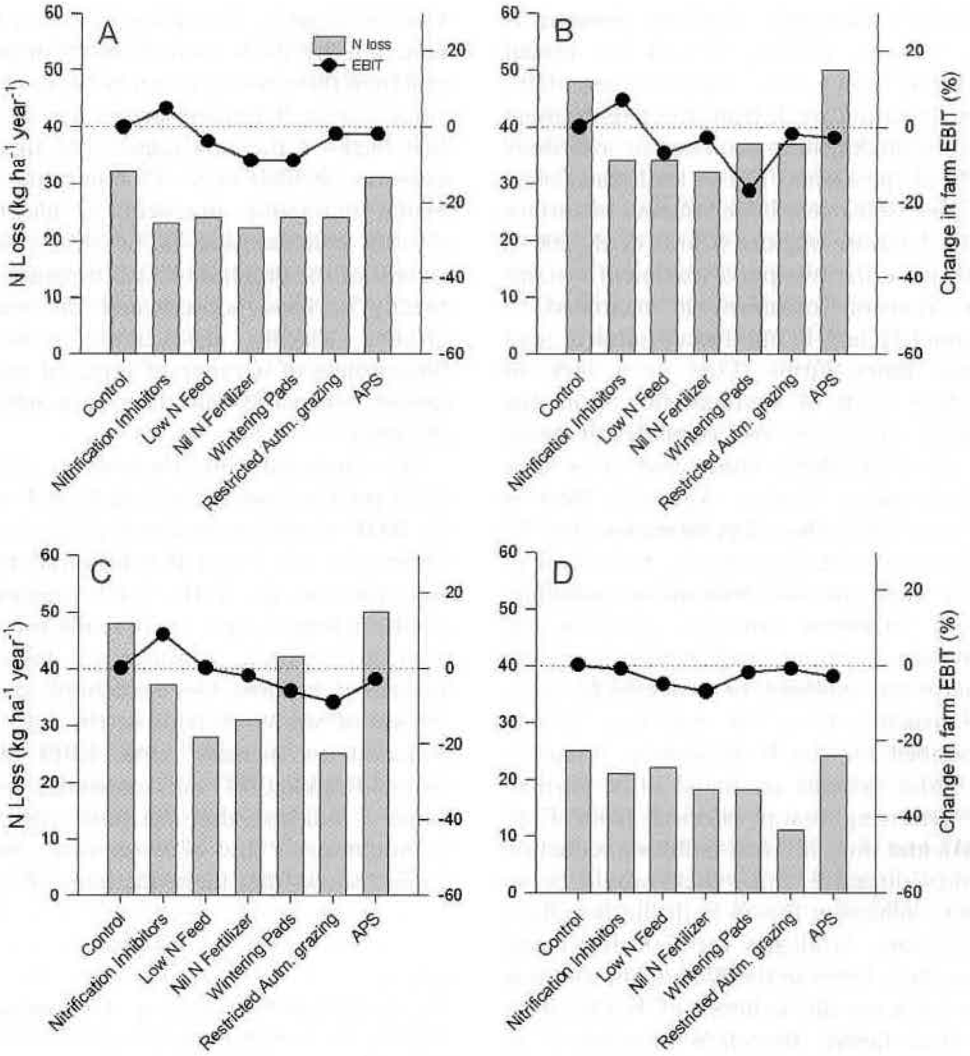


Fig. 5. Total N losses to waterways (bars) and projected cost-effectiveness (symbols) of a range of source management mitigation strategies which target N losses from dairy farms in the (A) Toenepi, (B) Waiokura, (C) Waikakahi and (D) Bog Burn catchments. Dashed line represents nil change in farm earnings before interest and tax (EBIT) values.

(Hunter et al., 1992; Vinten et al., 2004). The importance of overland flow is further supported by field measurements of *E. coli* losses in overland flow over a 3-year period from a grazed dairy pasture, typical of that found in the Bog Burn catchment (Monaghan et al., unpublished data). If we construct a simple inventory of these dairy farm sources by combing the mean *E. coli* concentration data documented in Table 4 with measured or estimated water yields for each source, we can identify, albeit at a relatively crude level, some "hot-spots" where mitigation technologies are best targeted. For simplicity, we have ignored the direct deposition of dung into streams as a source of faecal bacteria, given the high proportion of stream reaches on dairy farms that are fenced to exclude stock.

As noted for nutrient losses, this inventory analysis reveals marked contrasts in the major sources of faecal bacteria losses from dairy farms in the four catchments. In the case of the Bog Burn catchment, direct drainage of

Table 4
Reported or measured *E. coli* concentrations in identifiable flow sources from dairy farms

Source/pathway	Concentration cfu/100 ml	Reference
Direct drainage of un-treated effluent	5×10^6	Monaghan and Smith (2004)
Rain-fed mole-pipe drainage	4×10^3	Monaghan and Smith (2004) and unpublished data
Overland flow	3×10^4	R. Monaghan, unpublished data.
Soil matrix flow	2	Hunter et al. (1992)
Border dyke wash	4×10^4	Carey et al. (2004) and unpublished data.
Two-pond effluent treatment ponds	7×10^4	Hickey et al. (1989)
Advanced Pond treatment Systems	9×10^2	Craggs et al. (2004)

irrigated FDE through mole-pipe drainage systems is estimated to provide approximately 78% of the annual *E. coli* load transferred from soil to water on these dairy farms. In contrast, irrigation wash from the free-draining soils in the Waikakahi catchment is estimated to contribute approximately 98% of the annual *E. coli* load transferred from soil to water, due to the very large volumes of surface water lost under this land-use practice (Carey et al., 2004). Discharges of FDE from the two-pond treatment systems still used in the Toenepi catchment is estimated to contribute approximately half of the faecal bacteria load emitted from these dairy farms. Due to a lack of information regarding yields of overland flow from the soils in the Waiokura catchment, our estimates of faecal bacteria sources are particularly crude and it is thus difficult to pin-point major sources. Although there is much uncertainty attached to these flux estimates, they do guide us to some obvious mitigation options. In the case of the Bog Burn farms, these options would include adopting a deferred effluent irrigation strategy, installing an Advanced Pond System, applying small depths of effluent by increasing irrigator groundspeed, or switching from the current travelling irrigator to a low rate (e.g. K-line) applicator, as described for the P mitigations discussed above. Advanced Pond Systems are noted to be particularly effective at removing faecal bacteria from FDE (Craggs et al., 2004), and thus, if faecal pollution reduction was the major policy objective, this system would be an effective approach to achieving this in both the Bog Burn and Toenepi catchments. Strategies for reducing faecal bacteria losses from dairy farms in the Waikakahi would in contrast focus on reducing the volumes of border dyke wash lost from these farms. Irrigation bunding is an obvious measure to consider here, as well as ensuring irrigation clock timings are correctly set and irrigation headraces are well maintained to ensure even water delivery. The irrigation bunding in the Waikakahi catchment would have the combined effect of also reducing P losses (Fig. 4). Changing to spray irrigation delivery of water would also much reduce the transfer of faecal bacteria to waterways, although this would incur considerable extra infrastructure costs and energy requirements.

5. Prioritisation of mitigation measures: the importance of farm and catchment context

When considering management interventions for the control of farm pollutant losses, it is important to first clearly establish the need and urgency for reducing the loss of a specific pollutant. Experience informs us that some catchments are more sensitive to nutrient pollution than others, and some are specifically sensitive to P whilst others may be more sensitive to N. As an example, monitoring of the Oreti River catchment, which encompasses the Bog Burn study catchment described here, shows that nuisance weed and algal growth in this river system currently appears to be P- rather than N-limited (Crawford, 2001). As a consequence, farm mitigation measures which target P reduction are likely to have more impact at a catchment level than those which target N losses. This is an important consideration if farmers are to get the "biggest bang for their buck" if they are required to finance environmental measures on their farms (NZ farming, including environmental mitigation practices, is unsubsidised). Another obvious consideration is to clearly identify the major sources of the pollutant so as to ensure BMPs are targeted directly at these sources and the measure is therefore effective. Thirdly, an accurate assessment of the cost effectiveness of a range of targeted mitigation options is needed to ensure that farm expenditure is used most efficiently.

The projected cost-effectiveness, expressed as dollars saved per kilogram of reduced N or P emission, of each of the BMP measures evaluated above is shown in Table 5. Optimising soil Olsen P values and the use of nitrification inhibitors are BMPs that represent a financial saving in all catchments and are thus the most obvious management intervention measures to implement in the first instance if nutrient loss reductions are sought. Likewise, the use of wintering pads in the Bog Burn catchment is projected to increase farm EBIT whilst reducing N losses. Ranking of the remaining measures shown in Table 5 indicates that the most cost effective approach to minimising P and N losses varies between catchments. Table 5 shows that the reduction of P losses from Toenepi

Table 5
Cost effectiveness ($ saved per kg of nutrient conserved) of mitigation measures for reducing losses of N and P from dairy farms in the 4 catchments

	Toenepi	Waiokura	Waikakahi	Bog Burn
Phosphorus				
Optimum. Olsen P	113	221	421	490
Deferred effluent irrigation	−22	0	n/a	−44
Low rate effluent irrigation	−8	n/a	n/a	−21
Small effluent application depth	n/r	n/a	n/a	−24
Irrigation bunding	n/r	n/r	−15	n/r
Low solubility P fertiliser	n/a	n/a	0	0
APS	−108	n/a[a]	n/a[a]	n/a[a]
Nitrogen				
Nitrification inhibitors	10	11	16	5
Restricted autumn/winter grazing	−5	−5	−6	−1
Nil N fertiliser input	−16	−4	−1	−16
Low N feed	−12	−13	0	−41
Wintering pads	−24	−36	−9	2
APS	−20	n/a[a]	n/a[a]	−52

Figures are expressed relative to the control farm system in each catchment.
n/a = not applicable.
n/r = not relevant.
[a] Projected to increase P or N losses from these model farms due to change from land application of FDE to treatment via an APS.

farms would be most cost-effectively achieved by using a low rate application system for irrigating FDE to land out of the existing two-pond treatment system. In the Bog Burn catchment the most cost-effective measures are, in decreasing order of cost-effectiveness, (i) the use of a less soluble form of P fertiliser (although this is estimated to reduce farm P losses by only 5%), (ii) low rate irrigation of FDE, (iii) applying effluent in small application depths and (iv) using a deferred effluent irrigation management strategy. The most cost-effective measures for reducing N losses from dairy farms in the Bog Burn catchment are, in decreasing order of cost-effectiveness and assuming that nitrification inhibitors and wintering pads are already adopted as BMPs: (i) a restricted autumn grazing regime, (ii) reducing N fertiliser inputs, and (iii) importing low N feed supplements. This ranking contrasts with that observed for the other catchments, where restricted winter grazing (Toenepi), eliminating fertiliser N inputs (Waiokura) or importing low N feed supplements are the next most cost-effective N mitigation measures after nitrification inhibitors.

In addition to the cost-effectiveness data described in Table 5, farm context is also an important consideration when mitigation measures are being deliberated and prioritised (Bewsell and Kaine, 2005, 2006). Some changes to management systems require infrastructure that may not be present on the existing farm, or the additional infrastructure required may serve a useful secondary role that is of benefit to the farmer. There are at least two obvious examples of this relevant to the catchment case study farms evaluated here. The first is the introduction of a low rate application system for irrigating FDE to land in the Toenepi or Bog Burn catchments. In the case of the Toenepi farms, the two-pond system is already present, thus providing storage and solid separation functions that are beneficial to the operation of the low rate effluent management system. As a consequence, the main additional infrastructure required is limited to the irrigation pump and pipe-work, in contrast to the Bog Burn farms where effluent is typically pumped from a sump with very little effluent storage provided. A second example is the introduction of a wintering pad system to farms in the Bog Burn catchment. Although we have discussed the merits of this management system within the context of a water quality issue, the use of such pads has a number of other features that will influence farmers' adoption of this technology. Other positive features of wintering pads include an increased sense of control over stock wintering practices and thus stock health (for some farms this herd management aspect is often contracted out to other non-dairy farms over winter), and the availability of a pad system for grazing-off cows during wet spring periods when soil and pasture treading damage is a problem. On the other hand, the logistics of handling large volumes of effluent captured in the loafing areas of such pads adds complexity to the farm operation.

5.1. Implications for greenhouse gas emissions and energy use

The wider environmental consequences of introducing mitigation options is another important consideration for prioritising their on-farm adoption. Therefore, total greenhouse gas (GHG) emissions and energy use were also estimated for each mitigation option documented in Figs. 4 and 5. The P source mitigations did not have any impact on total GHG emissions from the case study farms. Due to a lack of data on methane (CH_4) and nitrous oxide (N_2O) emissions from Advanced Pond Systems, GHG emissions under this management scenario were assumed to be similar to those modelled for the average farms. With respect to the N source mitigations, the largest reductions were achieved with the nil N fertiliser option, where total GHG emissions from the case study farms were reduced by 17–31%. The largest reduction was achieved in the Waikakahi farm, which reflects the relatively high amount of N fertiliser use in the case-study farm in this catchment. The reductions in the other catchments were all around 17%. Although this mitigation option currently incurs a small reduction in farm profitability (Fig. 5), potential savings associated with a reduction in GHG emissions could tip the balance. Although there is currently no mechanism in New Zealand to accrue these potential savings, at an assumed value of \$25/t CO_2-equivalent, farm EBIT values are similar to those of the case study farms.

Total on-farm reductions in GHG emissions achieved under the low N feed and the nitrification inhibitor options were similar at 5–14% and 6–11%, respectively. Although the use of nitrification inhibitors was assumed to reduce direct emissions of nitrous oxide by 75% (de Klein et al., 2006), total on-farm reductions were much smaller due to an increase in both CH_4 and carbon dioxide (CO_2) emissions. Nitrification inhibitors do not directly affect emissions of these latter GHGs, but their use was estimated to result in an increase in pasture production, feed intake and thus milk production, therefore increasing CH_4 and CO_2 emissions. When expressed per unit of milk production, the use of nitrification inhibitors reduced GHG emissions by about 15% for all case study farms. The restricted autumn/winter grazing option did not have a substantial impact on the total GHG emission of the case study farms, with emissions being either reduced (North Island farms) or increased (South Island farms) by about 3%. The winter pad option increased total GHG emissions on all case study farms by 2–8%. In practice, these winter management options may reduce soil damage from animal treading which has been shown to decrease N_2O emissions.

In terms of energy use, our analyses showed that restricted autumn/winter grazing and nitrification inhibitor management interventions slightly increased total farm energy use by 1–3% and 3–7%, respectively. In the North Island catchments the winter pad option substantially increased energy use (by about 16%) due to the increased energy requirement for harvesting pasture and feeding out on the winter pad. In the South Island catchments, this increase in energy use was largely off-set by a reduction in energy use required for the transport of animals to and from the run-off wintering area and, more importantly, for cultivation of the off-farm winter forage crop. As a result, total farm system energy use under the winter pad option in the South Island catchments increased by only 1–3%. Large reductions in energy use were achieved under the low N input and low N feed options of 23–45% and 7–24%, respectively, due to the substantial reduction in the use of N fertiliser, which has a high energy cost associated with its production. These large reductions make the low N options less sensitive to any future increases in energy costs.

6. Summary

The on-going intensification of dairy farming systems is placing increased pressures on soil and water resources. As such, improved management systems that reduce the environmental impacts of this intensive land-use are continually sought, and with some urgency in regions where water bodies have been identified as particularly sensitive to further nutrient enrichment. Research efforts in the past 10–15 years have identified a range of management interventions that can reduce nutrient losses from dairy farms at source. Evaluation of these management mitigations shows that these options vary both in cost and effectiveness. Some management interventions, such as the use of nitrification inhibitors and limiting soil P fertility to economically optimal levels, can deliver win–win outcomes whereby nutrient losses are reduced whilst dairy farm profitability is improved. For many other interventions, however, a net cost is often incurred. In the case of many of the mitigations that target reductions in sources of P lost from dairy farms, these costs are relatively small and typically less than 4% of farm EBIT values. These measures are projected to deliver reductions in P loss from farms of between 28 (Bog Burn farms) and 52% (Waikakahi farms). Similar outcomes are evident for the implementation of a range of N mitigation strategies, although some are noted to incur greater cost than observed for the P mitigation strategies evaluated. Due to the contrasting physical resources and management systems present in the four dairy catchments evaluated here, management interventions need to be considered on a catchment-by-catchment and farm-by-farm basis.

As a concluding remark, we note that evaluation of the whole dairy system (i.e. dairy farm and associated land used for feed production) was an important scale of analysis when attempting to determine the cost-effectiveness of a range of mitigation options. This scale of analysis adds considerable complexity to the behaviour of the dairy system under evaluation. Because detection of experimental responses in replicated systems at this scale is very difficult to achieve, much reliance is thus placed on modelling tools that attempt to simulate system responses in response to a change in land-use management. This introduces at least two levels of uncertainty; the first concerns our understanding of how systems work and interact, and the second concerns the human factor of how farms are actually managed on a day-to-day basis. In terms of the former, based on our current state of knowledge, we currently have more confidence in model predictions of N losses from dairy farms than P, and in-turn considerably more confidence in P loss predictions than faecal bacteria. In terms of the human factor, uncertainty could be reduced by more rigorously surveying farmers' decision making processes regarding tasks such as effluent irrigation scheduling and fertilization timings.

Acknowledgements

The authors wish to thank the many farmers who provided helpful cooperation and regular feedback during the conduct of this catchments study. This work was supported by funding from Dairy Insight NZ, the Ministry of Agriculture and Forestry's Sustainable Farming Fund, the NZ Fertiliser Manufacturers' Research Association and the NZ Foundation for Research, Science and Technology (Contract C10 × 0320).

References

Bewsell, D., Kaine, G., 2005. Adoption of environmental best practice amongst dairy farmers. New Zealand Agricultural and Resource Economics Society Conference, 26–27 August, Nelson.

Bewsell, D., Kaine, G., 2006. Adoption of environmental best practice amongst dairy farmers. In: Australasia Pacific Extension Network International Conference, 6–8 March Beechworth, Australia.

Cameron, K.C., Di, H.J., McLaren, R.G., 1999. Comparison of nitrogen leaching losses from different forms and rates of organic wastes, fertilisers and animal urine applied to Templeton soil lysimeters. In: Currie, L.D., et al. (Eds.), Best Soil Management Practices for Production. Fertilizer and Lime Research Centre, Massey University, Palmerston North, New Zealand, pp. 241–249.

Carey, P.L., Drewry, J.J., Muirhead, R.W., Monaghan, R.M., 2004. Potential for nutrient and faecal bacteria losses from a dairy pasture under border dyke irrigation: a case study. Proceedings of the New Zealand Grassland Association 66, 141–149.

Clough, T.J., Sherlock, R.R., Cameron, K.C., Ledgard, S.F., 1996. Fate of urine nitrogen on mineral and peat soils in New Zealand. Plant and Soil 178, 141–152.

Craggs, R.J., Sukias, J.P., Tanner, C.T., Davies-Colley, R.J., 2004. Advanced pond system for dairy-farm effluent treatment. New Zealand Journal of Agricultural Research 47, 449–460.

Crawford, S., 2001. Water quality and our changing agricultural landscape. In: Proceedings of the Southland Dairy Forum, 2 April 2001. pp. 11–15.

Dexcel, 2003. Economic Survey of New Zealand Dairy Farmers, 2001–2002. Dexcel Limited, Hamilton, 40pp.

Di, H.J., Cameron, K.C., 2003. Mitigation of nitrous oxide emissions in spray-irrigated grazed grassland by treating the soil with dicyandiamide, a nitrification inhibitor. Soil Use and Management 19, 284–290.

Di, H.J., Cameron, K.C., 2005. Reducing environmental impacts of agriculture by using a fine particle suspension nitrification inhibitor to decrease nitrate leaching from grazed pastures. Agriculture, Ecosystems and Environment 109, 202–212.

Edmeades, D.C., Metherell, A.K., Waller, J.E., Roberts, A.H.C., Morton, J.D., 2006. Defining the relationships between pasture production and soil P and the development of a dynamic P model for New Zealand pastures: a review of recent developments. New Zealand Journal of Agricultural Research 49, 207–222.

Fonterra Co-operative Group, Local Government New Zealand (LGNZ), Ministry for the Environment (MFE), Ministry of Agriculture and Forestry (MAF), 2003. Dairying and Clean Streams Accord Availble from: 〈http://www.mfe.govt.nz/issues/land/rural/dairying-accord-may03.pdf〉 (accessed 03.07.06).

Fraser, P.M., Cameron, K.C., Sherlock, R.R., 1994. Lysimeter study of the fate of nitrogen in animal urine returns to irrigated pasture. European Journal of Soil Science 45, 439–447.

Gillingham, A.G., Thorrold, B.S., 2000. A review of New Zealand research measuring phosphorus in runoff from pasture. Journal of Environmental Quality 29, 88–96.

Hamill, K.D., McBride, G.B., 2003. River water quality trends and increased dairying in Southland, New Zealand. New Zealand Journal of Marine and Freshwater Research 37, 323–332.

Hart, R.P.S., Larcombe, M.T., Sherlock, R.A., Smith, L.A., 1998. Optimisation techniques for a computer simulation of a pastoral dairy farm. Computers and Electronics in Agriculture 19, 129–153.

Hickey, C.W., Quinn, J.M., Davies-Colley, R.J., 1989. Effluent characteristics of dairy shed oxidation ponds and their potential impacts on rivers. New Zealand Journal of Marine and Freshwater Research 23, 569–584.

Houlbrooke, D.J., Horne, D.J., Hedley, M.J., Hanly, J.A., Scotter, D.R., Snow, V.O., 2004a. Minimising surface water pollution resulting from farm-dairy effluent application to mole-pipe drained soils. I. An evaluation of the deferred irrigation system for sustainable land treatment in the Manawatu. New Zealand Journal of Agricultural Research 47, 405–415.

Houlbrooke, D.J., Horne, D.J., Hedley, M.J., Hanly, J.A., Snow, V.O., 2004b. A review of literature on the land treatment of farm-dairy effluent in New Zealand and its impact on water quality. New Zealand Journal of Agricultural Research 47, 499–511.

Hunter, C., McDonald, A., Beven, K., 1992. Input of fecal coliform bacteria to an upland stream channel in the Yorkshire Dales. Water Resources Research 28, 1869–1876.

de Klein, C.A.M., 2005. Mitigating the Environmental Impacts of Dairy Farming in NZ—Inventory of Research Efforts and Currently Available Mitigation Strategies. AgResearch Limited, Hamilton, New Zealand, 48pp.

de Klein, C.A.M., Ledgard, S.F., Clark, H., 2002. Evaluation of two potential on-farm measures for reducing greenhouse gas emissions from an average dairy farm on the West Coast of the South Island of New Zealand. Proceedings of the New Zealand Grassland Association 64, 159–165.

de Klein, C.A.M., Smith, L.C., Monaghan, R.M., 2006. Restricted autumn grazing to reduce nitrous oxide emissions from dairy pastures in Southland, New Zealand. Agriculture, Ecosystems and Environment 112, 192–199.

Larcombe, M.T., 1999. UDDER for Windows: a Desktop Dairyfarm for Extension and Research-Operating Manual. Maffra Herd Improvement Co-op, Maffra, Victoria, Australia.

Larned, S.T., Scarsbrook, M.R., Snelder, T.H., Norton, N.J., Biggs, B.J.F., 2004. Water quality in low-elevation streams and rivers of New Zealand: recent state and trends in contrasting land-cover classes. New Zealand Journal of Marine and Freshwater Research 38, 347–366.

Ledgard, S.F., Menneer, J.C., 2005. Nitrate leaching in grazing systems and management strategies to reduce losses. In: Currie, L.D., Hanly, J.A. (Eds.), Developments in fertiliser application technologies and nutrient management. Occasional Report No. 18. Fertiliser and Lime Research Centre, Massey University, Palmerston North, New Zealand, pp. 79–92.

Ledgard, S.F., Penno, J.W., Sprosen, M.S., 1999. Nitrogen inputs and losses from clover/grass pastures grazed by dairy cows, as affected by nitrogen fertilizer application. Journal of Agricultural Science 132, 215–225.

Ledgard, S.F., Finlayson, J.D., Gavin, J., Blackwell, M.D., Carran, R.A., Wedderburn, M.E., Jollands, N.A., 2003. Resource use efficiency and environmental emissions from an average Waikato dairy farm, and impacts of intensification using nitrogen fertiliser or maize silage. Proceedings of the New Zealand Grassland Association 65, 185–189.

Ledgard, S.F., Finlayson, J.D., Patterson, M.G., Carran, R.A., Wedderburn, M.E., 2004. Effects of intensification of dairy farming in New Zealand on whole-system resource use efficiency and environmental emissions. Life cycle assessment in the agri-food sector. In: Halberg, N. (Ed.), Proceedings of the 4th International Conference. Danish Institute of Agricultural Sciences, Report No. 61, Denmark, pp. 226–229.

LIC, 2005. Dairy Statistics. Livestock Improvement Corporation, NZ Availble from: 〈http://www.lic.co.nz/pdf/dairy_stats/Dairy_Statistics_05_LEE666.pdf〉.

Longhurst, R.D., O'Connor, M.B., Bremner, K., Matthews, L., 2000. Animal constraints to pasture treated with farm dairy effluent: preference under grazing and issues of faecal contamination. New Zealand Journal of Agricultural Research 43, 501–507.

MAF, 2003. Dairy Monitoring Report. Ministry of Agriculture and Forestry Availble from: 〈http://www.maf.govt.nz/mafnet/rural-nz/statistics-and-forecasts/farm-monitoring/index.htm〉.

McDowell, R.W., Catto, W., 2005. Alternative fertilisers and management to decrease incidental phosphorus loss. Environmental Chemistry Letters 2, 169–174.

McDowell, R.W., Monaghan, R.M., Morton, J., 2003a. Soil phosphorus concentrations to minimise potential P loss to surface waters in Southland. New Zealand Journal of Agricultural Research 46, 239–253.

McDowell, R.W., Monaghan, R.M., Carey, P.L., 2003b. Potential phosphorus losses in overland flow from pastoral soils receiving long-term applications of either superphosphate or reactive phosphate rock. New Zealand Journal of Agricultural Research 46, 329–337.

McDowell, R.W., Monaghan, R.M., Wheeler, D., 2005. Modelling phosphorus losses from pastoral farming systems in New Zealand. New Zealand Journal of Agricultural Research 48, 131–141.

McLeod, M., Aislabie, J.M., Ryburn, J., McGill, A., Taylor, M., 2003. Microbial and chemical tracer movement through two Southland soils, New Zealand. Australian Journal of Soil Research 41, 1163–1169.

McLeod, M., Aislabie, J., Ryburn, J., McGill, A., 2004. Microbial and chemical tracer movement through Granular, Ultic and Recent soils. New Zealand Journal of Agricultural Research 47, 557–563.

Monaghan, R.M., Smith, L.C., 2004. Minimising surface water pollution resulting from farm-dairy effluent application to mole-pipe drained soils. II. The contribution of preferential flow of effluent to whole-farm pollutant losses in subsurface drainage from a West Otago dairy farm. New Zealand Journal of Agricultural Research 47, 417–428.

Monaghan, R.M., Smeaton, D., Hyslop, M.G., Stevens, D.R., de Klein, C.A.M., Smith, L.C., Drewry, J.J., Thorrold, B.S., 2004. A desktop evaluation of the environmental and economic performance of model dairy farming systems within four New Zealand catchments. Proceedings of the New Zealand Grassland Association 66, 57–67.

Monaghan, R.M., de Klein, C.A.M., Wilcock, R.J., Smith, L.C., Thorrold, B.S., 2005a. Managing nutrient losses and greenhouse gas emissions from dairy farms within the Bog Burn catchment, Southland. In: Currie, L.D.x, Hanly, J.A. (Eds.), Developments in Fertiliser Application Technologies and Nutrient Management. Occasional Report No. 18. Fertiliser and Lime Research Centre, Massey University, Palmerston North, New Zealand, pp. 151–161.

Monaghan, R.M., Paton, R.J., Smith, L.C., Drewry, J.J., Littlejohn, R.P., 2005b. The impacts of nitrogen fertilisation and increased stocking rate on pasture yield, soil physical condition and nutrient losses in drainage from a cattle-grazed pasture. New Zealand Journal of Agricultural Research 48, 227–240.

Morton, J.D., McDowell, R.W., Monaghan, R.M., Roberts, A.H.C., 2003. Balancing phosphorus requirements for milk production and water quality. Proceedings of the New Zealand Grassland Association 65, 111–115.

NZFMRA, 2002. Code of Practice for Fertiliser Use. New Zealand Fertiliser Manufactures' Research Association, Auckland, New Zealand Availble from: 〈http://www.fertresearch.org.nz/fertiliser.cfm?folderid = 40andsectionid = 18andparentid = 18〉.

PCE, 2004. Growing for Good. Parliamentary Commissioner for the Environment, Wellington, New Zealand Availble from: 〈http://www.pce.govt.nz/reports/allreports/1_877274_45_3.shtml〉.

Roach, C.G., Longhurst, R.D., Ledgard, S.F., 2001. Land application of farm dairy effluent for sustainable dairy farming. Proceedings of the New Zealand Grassland Association 63, 53–57.

Scholefield, D., Tyson, K.C., Garwood, E.A., Armstrong, A.C., Hawkins, J., Stone, A.C., 1993. Nitrate leaching from grazed grassland lysimeters: effects of fertilizer input, field drainage, age of sward and patterns of weather. Journal of Soil Science 44, 601–613.

Sharpley, A.N., Syers, J.K., 1979. Loss of nitrogen and phosphorus in tile drainage as influenced by urea application and grazing animals. New Zealand Journal of Agricultural Research 22, 127–131.

Silva, R.G., Cameron, K.C., Di, H.J., Hendry, T., 1999. A lysimeter study of the impact of cow urine, dairy shed effluent, and nitrogen fertiliser on nitrate leaching. Australian Journal of Soil Research 37, 357–369.

Thorrold, B.S., Rodda, H., Monaghan, R.M., 1998. Modelling landuse effects on water quality in the Oteramika catchment. Client Report to Southland Regional Council on the Oteramika Trial Catchment Project. AgResearch, Hamilton, New Zealand, 47pp.

Vant, W.N., 2001. New challenges for the management of plant nutrients and pathogens in the Waikato River, New Zealand. Water Science and Technology 43 (5), 137–144.

Vinten, A.J.A., Lewis, D.R., McGechan, M., Duncan, A., Aitken, M., Hill, C., Crawford, C., 2004. Predicting the effect of livestock inputs of *E. coli* on microbiological compliance of bathing waters. Water Research 38, 3215–3224.

Wells, C., 2001. Total energy indicators of agricultural sustainability: Dairy farming case study. Report to the New Zealand Ministry of Agriculture and Forestry, 79pp.

Wheeler, D.M., Ledgard, S.F., de Klein, C.A.M., Monaghan, R.M., Carey, P.L., McDowell, R.W., Johns, K.L., 2003. OVERSEER® nutrient budgets 2-moving towards on-farm resource accounting. Proceedings of the New Zealand Grasslands Association 65, 191–194.

Wilcock, R.J., Nagels, J.W., Rodda, H.J.E., O'Connor, M.J., Thorrold, B.S., Barnett, J.W., 1999. Water quality of a lowland stream in a New Zealand dairy farming catchment. New Zealand Journal of Marine and Freshwater Research 33, 683–696.

Wilcock, R.J., Monaghan, R.M., Quinn, J.M., Campbell, A.M., Duncan, M.J., Thorrold, B.S., McGowan, A.W., Betteridge, K., 2006. Land use impacts and water quality targets in the intensive dairying catchment of the Toenepi Stream, New Zealand. New Zealand Journal of Marine and Freshwater Research 40, 123–140.

ELSEVIER

Journal of Environmental Management 87 (2008) 623–632

Journal of
Environmental
Management

www.elsevier.com/locate/jenvman

Review

Development of a risk-based index for source water protection planning, which supports the reduction of pathogens from agricultural activity entering water resources

Michael Goss*, Charlene Richards

Centre for Land and Water Stewardship, University of Guelph, Kemptville Campus, 831 Prescott Street, Kemptville, Ontario, K0G 1J0, Canada

Received 12 March 2006; received in revised form 14 December 2006; accepted 14 December 2006

Abstract

Source water protection planning (SWPP) is an approach to prevent contamination of ground and surface water in watersheds where these resources may be abstracted for drinking or used for recreation. For SWPP the hazards within a watershed that could contribute to water contamination are identified together with the pathways that link them to the water resource. In rural areas, farms are significant potential sources of pathogens. A risk-based index can be used to support the assessment of the potential for contamination following guidelines on safety and operational efficacy of processes and practices developed as beneficial approaches to agricultural land management. Evaluation of the health risk for a target population requires knowledge of the strength of the hazard with respect to the pathogen load (mass × concentration). Manure handling and on-site wastewater treatment systems form the most important hazards, and both can comprise confined and unconfined source elements. There is also a need to understand the modification of pathogen numbers (attenuation) together with characteristics of the established pathways (surface or subsurface), which allow the movement of the contaminant species from a source to a receptor (water source). Many practices for manure management have not been fully evaluated for their impact on pathogen survival and transport in the environment. A key component is the identification of potential pathways of contaminant transport. This requires the development of a suitable digital elevation model of the watershed for surface movement and information on local groundwater aquifer systems for subsurface flows. Both require detailed soils and geological information. The pathways to surface and groundwater resources can then be identified. Details of land management, farm management practices (including animal and manure management) and agronomic practices have to be obtained, possibly from questionnaires completed by each producer within the watershed. To confirm that potential pathways are active requires some microbial source tracking. One possibility is to identify the molecular types of *Escherichia coli* present in each hazard on a farm. An essential part of any such index is the identification of mitigation strategies and practices that can reduce the magnitude of the hazard or block open pathways.

Keywords: Source water protection; Risk index; Manure; On-site waste treatment systems; Microbial pathogens; Persistence; Transport; Attenuation; Pathways; Barriers

1. Introduction

Source water protection planning is an approach to prevent contamination of ground and surface water in watersheds. Contaminated potable and recreational waters pose a direct threat to human health. Indirect threats to health arise where water is used to irrigate fresh market crops. Evaluation of the health risk from a water resource (Collins and Rutherford, 2004; Gale, 2005) requires knowledge of the strength (pathogen load) of the hazard, and understanding the modification of pathogen numbers (attenuation) together with characteristics of the transport pathways (surface or subsurface).

Animal production and feeding operations represent a significant potential source of pathogens (Center for Disease Control and Prevention, 1998; Duffy, 2003; Gessel et al., 2004; Gerba and Smith, 2005). Pathogens of greatest

---

*Corresponding author. Tel.: +1 613 258 8336x333; fax: +1 613 258 8477.

E-mail addresses: mgoss@uoguelph.ca (M. Goss), crichard@uoguelph.ca (C. Richards).

0301-4797/$ - see front matter © 2007 Elsevier Ltd. All rights reserved.
doi:10.1016/j.jenvman.2006.12.048

Table 1
Characteristics of diseases caused by pathogens that can originate from farm waste (based on CAST, 1994)

Pathogens	Disease severity	Fatality case rate (%)	No. of cells to cause illness
E. coli 0157:H7	Moderate–severe	2.0	$<10^3$–10^9 CFU
Salmonella	Mild–severe	0.1	1–10^9 CFU
Campylobacter	Mild–Moderate	0.05	500 CFU
Cryptosporidium	Moderate–severe		10–30 Oocysts
Giardia	Mild–Moderate		10 Cysts

concern are shed into the environment in great numbers, are highly infectious to humans and other animals at relatively small doses (Table 1), survive and remain infectious in the environment for a considerable time (Avery et al., 2005; Kearney et al., 1993; Hutchinson et al., 2004), tend to be resistant to water treatment (Erlandsen and Mayer, 1984) and thereby increase water treatment costs. Infectious, water-related diseases are a major cause of morbidity, with an estimated 1.6 million deaths worldwide (WHO, 2003). Contaminating species include the parasite *Cryptosporidium parvum*, viruses and bacterial pathogens. In May of 2000 the town of Walkerton, Ontario, experienced the largest waterborne disease outbreak in Canada. The town water system was contaminated with pathogens that originated in manure and resulted in 2300 cases of gastroenteritis and seven deaths (O'Connor, 2002). Despite a thorough judicial inquiry, the actual pathway between the possible microbial sources and the groundwater could not be confirmed. Pathogen die-off in soil is potentially huge (reduction $>10^{20}$) and therefore soil represents a major barrier that helps to protect water resources (Gale, 2005). However, soil processes can be compromised and there is a need to develop indicator frameworks that allow semi-quantitative risk assessments to be made for pathogens until information accrues that will permit a full quantitative analysis. An essential part of any index is the identification of mitigation strategies and practices that can reduce the magnitude of the hazard or block open pathways between source and water resource. This paper describes the sources of pathogen contamination that can exist on farms and the processes that help attenuate pathogens and protect water resources.

2. Agricultural sources of pathogens (hazards)

Sources of faecal contaminate are diverse, and include confined and distributed sources (Joel and Karns, 2000). Distributed or unconfined sources include faeces deposited by pastured animals and wildlife, together with recognised organic amendments, such as manure and sewage biosolids. Partially confined sources include animal feedlots, animal housing and manure storages, and on-site wastewater treatment systems (Wyer et al., 1996; Jones and Obiri-Danso, 1999). Many sources are subject to some level of control depending on decisions of the farmer.

Pathogens in manure, septage and sewage biosolids include *Listeria*, *Campylobacter*, *Salmonella*, *Escherichia coli (E. coli)* O157:H7, *Cryptosporidium* and *Giardia* (Hinton and Bale, 1991; Mawdsley et al., 1995; Pell, 1997; Wallis et al., 1996). Animal manure may contain disease organisms that are particular to the animal group but can also contain zoonotic disease organisms. Organic wastes become contaminated with human disease organisms coming from wild animals or because of failure in the segregation of waste streams. A small concentration of disease organisms relative to the total number of microbes has resulted in the use of indicator organisms, which provides an early warning that human pathogens may also be present. The enteric bacterium, *E. coli*, represents a reasonably effective indicator of the microbial quality of municipal drinking water (Medema et al., 1997) and domestic rural wells (Raina et al., 1999). *E. coli* concentrations of 10^8 to 10^{10} cfu per gram dry weight of freshly excreted faeces are three or more orders of magnitude greater than the upper values reported for pathogenic species.

A number of factors, including health status, animal type, age, diet, stress level and season determine the pathogen shedding rate (Joel and Karns, 2000; Nicholson et al., 2005; Russell et al., 2000; Bach et al., 2005a). The number of animals carrying or shedding pathogens at any one time varies in and between herds. Antibiotic treatment can induce shedding of infective material in some organisms (e.g. *E. coli* O157:H7) (Gyles, 2000), while a change of diet and feed regimes can have the same effect (Russell et al., 2000). Animals carrying zoonoses, such as *E. coli* O157:H7 may be symptom-free, so producers may be unaware that a problem exists. Pathogen concentrations in faeces may be reduced by employing probiotics (Fuller, 1999), vaccine technology (Glass, 2004) or the use of bacteriophages specific to an individual species or strain (Raya et al., 2006).

2.1. *Confined and semi-confined sources*

Domestic wastewater in rural areas is often treated by septic or other on-site wastewater disposal systems (OSWDS). Cleaning water from milking parlours and food processing facilities may be handled in the same way. Contaminants are initially confined in a tank, which retains solids and allows primary digestion to take place, while water and soluble materials seep into the soil from a distribution system.

Many outbreaks of waterborne decease can be traced to improperly functioning or poorly positioned OSWDS (Hagedorn et al., 1981). The capacity of soil to absorb effluent water is an important property; too rapid a flow can allow significant numbers of pathogens to move to groundwater (Yates, 1985). OSWDS need appropriate soil characteristics, topography and horizontal distance to water courses to be effective in minimizing contamination (Day, 2004). Critical design parameters therefore include the depth to the water table or to bedrock and permeability of the subsurface soil, with wet soils (or those subject to inundation) or steeply sloping sites enhancing the likelihood that open pathways for transport will be present.

Key potential sources of pathogens on farms are those associated with the management of animal manure. Manure management can directly control pathogen load and indirectly influence survival and transport of pathogens from the soil to water resources through modifying the microbial environment. Housing animals and poultry usually requires some temporary storage of manure within the barn, before transfer to longer-term storage in readiness for land application. Faeces from livestock in outside pens or corrals are only partly confined unless runoff water is collected.

Storage changes bacterial populations. A slurry store or solid manure heap is likely to consist of excreta of different ages and may even come from different barns. Rates of pathogen decline in manures can be affected by diet, which also determines both physical and chemical properties of faecal and storage conditions (Plachá et al., 2001). Nicholson et al. (2005) concluded that temperature, aeration, pH and dry matter content, determine pathogen declination rates during storage. However, many of these factors vary with management practices. *Cryptosporidium parvum* oocysts survived in stored slurry despite the high levels of ammonium (Fleming et al., 1997). *Giardia* appears to be sensitive to freezing, whereas survival of other pathogens is enhanced. Temperatures above 30 °C generally reduce survival times and few organisms appear to survive for long in dried manure (Table 2).

Cattle slurry and poultry excreta contain concentrations typically about ten times greater than in pig slurry (Nodar et al., 1990). Initially, populations of viable organisms decline abruptly (Nodar et al., 1992) although persistence of *E. coli* O157:H7 (Bach et al., 2005b) can be more than 30 days at 22–23 °C, and even longer at lower temperatures (Kudva et al., 1998). *E. coli* O157:H7 survived for 21 months in an outside manure pile stored under fluctuating environmental conditions; survival was shorter in slurries and organisms were undetectable after 5 days incubation at 23 °C (Kudva et al., 1998).

Manure treatment on farms can be through composting, both in-vessel and in windrows, which requires that a temperature above 55 °C be maintained long enough to kill pathogens (St. Jean, 1997). Mechanical separation of coarse solids from slurry produces a material that can be stacked and composted, with liquid being treated independently. *Cryptosporidium* oocysts introduced to residual liquid separated from cattle slurry became non-viable after 4.1 days (Read and Svoboda, 1995). Anaerobic digesters are in use, where the temperature for the process is either at ambient, when bacteria are not killed, or at temperatures of at least 55 °C, when pathogens are killed. Survival of 10% of *E. coli* and *Campylobacter jejuni* (*C. jejuni*) for longer than 50 days (Kearney et al., 1993), and for bovine enterovirus, longer than 13 days (Monteith et al., 1996), occurred at temperatures below 40 °C. Lime treatment (addition of quick or slaked lime) to raise the pH to 12 for at least 2 h (Table 3) has also been used for septage.

Structural reliability of containers for manure or domestic wastewater needs to be included in the risk assessment. The majority of storage systems are open-topped, so they collect precipitation but allow free gaseous exchange. Poor maintenance and earthen storages can lead to groundwater pollution (Rowsell et al., 1985;

Table 3

Effect of treatment on the survival of bacteria (based on Millner, P., Personal Communication, March 2003)

Treatment	Log reduction	Stress
Lagoon	1–3	Time
Constructed wetland	2–3	Time, filtration
Deep stack (composting)	1–?	NH_3, heat
Digestion – mesophilic	1–2	Time; heat
Digestion – thermophilic	5	
Composting	1–5	Heat, time
Air drying	1–2	Dessication
Heat drying	4–5	Heat, dessication
Pasteurization	5	Time, heat
Alkaline process	3–5	Heat, NH_3

Table 2

Persistence of potentially pathogenic organisms in manure (based on Wang et al., 1996; Jiang et al., 2002; Bach et al., 2005a)

Organism	Persistence under experimental conditions (days)					
	Frozen	5 °C	30 °C	Liquid manure	Compost	Dried
E. coli	>100	>100	10	100	7	1
Salmonella	>150	150	28	75	14	7
Campylobacter	50	21	7	100	7	1
Giardia	<1	7	7	300	14	1
Cryptosporidium	>300	50	28	>300	28	1
E. coli O157:H7		~5 °C 70	~15 °C >226	~22 °C 231		

Barrington et al., 1991). Earthen storage, in areas with shallow bedrock, pervious soils, and shallow water tables endanger water supplies unless artificial liners are used (Barrington et al., 1991). Problems with liquid manure storage systems contributed 17% of 229 listed manure spills recorded in Ontario (Blackie, M. Personal Communication, December, 2000).

Storage volume is an important issue: 34 of 38 manure spills were associated with problems from stored manure in the Southwestern Region of Ontario between 1988 and 1999. Limited storage (e.g., <180 days) increases the risk of spreading during conditions that can lead to environmental contamination (Fleming and Fraser, 2000). Spillage around stores represents a further source of contamination (Rudolph, 2003).

2.2. *Unconfined sources*

Direct animal access to water bodies poses a clear risk of contamination (Gary et al., 1983), which can be lessened by providing alternative drinking points. Behaviour of livestock is also important (Duncan et al., 1998; Veira et al., 2003). Direct voiding of faeces into streams can increase survival rates (Davies et al., 1995) as rapid sorption onto bed sediment can occur (Whiteley, 1998). Nevertheless, bacterial concentrations of $10\,\mathrm{mL}^{-1}$ have been found 20 km downstream from a source of contamination (Feresu and Van Sickle, 1990). Access to streams also allows animals to disturb the sediment, causing the release of stored pathogens into the water.

The majority of contamination incidents of water courses in the Southwestern Region of Ontario between 1988 and 1999 were related to land application of manure (Blackie, M. Personal Communication, 2000). Liquid manure is transported from storage to the field via pipelines, tanker-trailers, or custom truck-spreaders. There are three main methods of applying manure and biosolids: broadcasting, irrigation, and injection. Application method is important for the potential movement of pathogens. Liquid manure is applied to the soil surface of arable land or injected below the soil surface. The latter reduces the likelihood of any pathogens becoming airborne during spreading. Broadcast application of liquid manure from a tanker has resulted in fewer than a third of the problems of surface water contamination encountered with the use of spray irrigation. Failure of equipment associated with the land application caused 27% of manure spills that resulted in contamination of water courses in the South Western Region of Ontario between 1988 and 1999 (Blackie, M. Personal Communication, December, 2000).

Solid manure is typically applied by surface spreading, which then requires a second tillage operation for incorporation. Crane et al. (1983) concluded that land application of organic waste can significantly increase bacteria contamination of surface water from runoff, especially if farmers do not follow wise management options and safety precautions. Results from the *Ontario Farm Groundwater Quality Survey* indicated that farmstead drinking water wells were more likely to be contaminated where manure was spread (Goss et al., 1998; Rudolph et al., 1998).

Factors to consider in relation to timing of application include the risks from soil compaction, likelihood of runoff and nutrient loss. Limited manure storage on many farms means the common periods for application are the fall, winter, and spring. Spring application may be limited by soil wetness, while fall or winter applications tend to result in longer survival times. The likelihood of bacteria moving into water resources declines with time after manure application due to die off, which takes longer in manure applied in late fall, shortly before any freeze-up. Application as a side dressing for maize (*Zea mays* L.) (generally in mid-June, when soils are relatively dry and warm) results in the shortest period of survival. In the first day after applying liquid manure, more bacteria may be lost in overland flow from no-till land than from ploughed land, but the rate of decline in the concentration of bacteria in the runoff water can also be greater (King et al., 1994).

The most frequently reported route by which liquid manure can contaminate surface water courses is outflow from tile-drain systems (e.g., Fleming and Bradshaw, 1991, 1992a, b). After liquid manure application, bacteria move rapidly to the tile drains, particularly if soils are close to field capacity or after injection (Foran et al., 1993). Of the manure spreading events investigated by Dean and Foran (1991), 75% resulted in water quality impairment. It is difficult to determine an acceptable rate of liquid manure application, due to the numerous factors which influence the potential for contamination of watercourses (Foran et al., 1993). Routine inspection of tile drain outfalls during manure application could reduce adverse impacts on water quality. Pre-tillage of soil before spreading liquid manure can disrupt pore continuity and minimize the direct impacts on tile-drain quality (Fleming and Bradshaw, 1992b).

3. Persistence of pathogens in the environment

A number of factors influence the survival of pathogens following land application of manure: properties of the soil, availability of nutrients (including carbon) and interactions with soil biota (Abu-Ashour et al., 1994). These interactions include competition with and predation by indigenous soil micro- and meso-organisms. The application medium also affects the survival of bacteria. For example, Östling and Lindgren (1991) found that 20–40 times more indigenous Bacillus spores were present on manured crops than on un-manured crops, and these numbers remained constant with time to harvest. However, bacteria originating in the manure itself, such as *Clostridium*, some coliforms, and *E. coli*, all declined with time after manure application. Thelin and Gifford (1983) showed that when freshly voided manure was subject to rainfall simulation within 5 days, the concentration of faecal coliform bacteria in runoff was in the order of $10^4\,mL^{-1}$, but this number declined to $400\,mL^{-1}$ after 30 days.

Using four undisturbed 50 cm clay loam cores, Unc (2002) demonstrated that up to 80% of the bacteria applied with liquid swine manure was recovered in the drainage water. On the other hand a significant proportion of the faecal bacteria applied with the solid beef manure were retained within the manure matrix.

Once in surface water, the survival of *E. coli* (ETEC), *C. jejuni* and *Yersinia enterocolitica* is such that this can be a persistent venue for transmission between animals and humans (Terzieva and McFeters, 1991). However, the survival of pathogens in water and soil is very variable (Feachem et al., 1983) and differs between species. Among bacteria, many Gram positive organisms form resistant spores, whereas in Gram negative organisms the physiological adaptation to environmental stress may involve the reduction in cell size and metabolic rate (Roszak and Colwell, 1987). Some pathogenic bacteria can survive in a dormant state in soils and water but cannot be grown in conventional media (i.e. viable but non-culturable) (Byrd et al., 1991).

Soil parameters that modify survival of non-indigenous bacteria and viruses include soil pH, soil water content, organic matter content, soil texture, temperature, availability of nutrients, adsorption properties of the soil (Gerba et al., 1975; MacLean, 1983; Cools et al., 2001). Bacteria can survive longer in cold soils and fine textured soils (Fenlon et al., 2000). In cold soils ($<5\,°C$) *E. coli* can survive for up to 100 days. Survival of *Enterococcus spp.* was longer than that of *E. coli* at 5 °C, while the opposite was true at 15 °C and 25 °C (Cools et al., 2001). However, these authors also reported that while *Enterococcus spp.* survived longer in loamy soils than sandy soils at 25 °C, the reverse was true for *E. coli.* Gagliardi and Karns (2000) reported that *E. coli* O157:H7 was able to replicate in and migrate through cores of various soil types. Numbers of the pathogen in leachate correlated with ammonium and nitrate levels, and the numbers exceeded inoculum levels in all treatments (i.e. soil types, tilled and no till, and rainfall amounts) except in intact clay loam cores (Gagliardi and Karns, 2000). Survival of non-pathogenic *E. coli* exceeded 60 days at 25 °C and 100 days at 4 °C (Bogosian et al., 1996) and may be extended beyond that by residing within soil protozoa (Barker et al., 1999).

It is likely that at least part of the effect of soil texture is related to the water-holding capacity (Sadovski et al., 1978). Cools et al. (2001) found that the best survival of *E. coli* and *Enterococcus spp.* occurred in soils close to field capacity. Mubiru et al. (2000) showed that at the same gravimetric water content, the matric potential was lower in a silt loam with a clay content of $0.25\,g\,g^{-1}$ than one with $0.12\,g\,g^{-1}$, and the survival times for *E. coli* and *E. coli* O157:H7 were also shorter. *Campylobacter* species appear to have somewhat shorter survival times than *E. coli. C. jejuni* survived in soil for at least ten days but this number could double when the ambient temperature decreased to 6 °C (Lindenstruth and Ward, 1948).

Sunlight reduces the survival of bacteria and viruses in soil directly through the effect of UV light and as a result of drying (Yeager and O'Brien, 1979; Sinton et al., 1999; Gerba and Bitton, 1984; Mubiru et al., 2000). *Campylobacter* remains viable in a range of environments at 4 °C for up to 7 days (HMSO, 1993). Other environmental constraints include its inability to tolerate desiccation, low pH ($<$pH 5), exposure to O_2 and UV light (HMSO, 1993). About 100 days is the longest survival time reported for enteric viruses.

The survival potential of *E. coli* in the absence of water stress was shown to be at least 200 days and was not dependent on the initial concentration of *E. coli* in the applied manure (Unc and Goss, 2006a). Solid beef manure accelerated microbial activity immediately after application and increased the initial number of *E. coli*, but also shortened the survival length compared to liquid swine manure (Fig. 1). Solid manure led to decreased survival at all incubation temperatures considered including freezing. The impact of liquid manure on bacterial survival following freezing seems to vary with soil type (Fig. 2).

After 48 h, a population of *Salmonella*, introduced to soil containing earthworms, was reduced by a factor of four compared with *Salmonella* in a worm-free soil. Earthworms also caused a small reduction in the population of the indigenous soil bacteria. Free living protozoa, nematodes, and the soil bacterium *Bdellovibrio* are also predators of bacteria in the soil (Peterson and Ward, 1989). On average, 10% of faecal coliforms and faecal streptococci were still present in the soil 11 and 14 days respectively after application of pig manure (Chandler et al., 1981).

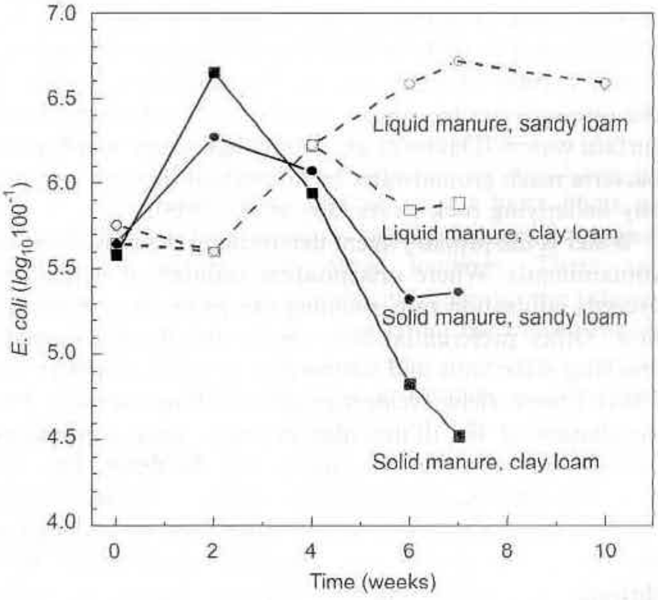


Fig. 1. Persistence of *E. coli* from manure after incorporation in soil (Redrawn from Unc and Goss, 2006a).

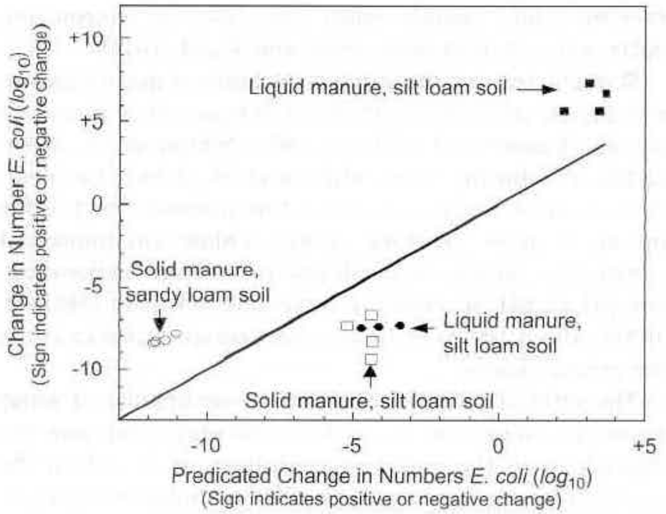


Fig. 2. Changes in numbers of *E. coli* ($\log_{10}100\,g^{-1}$ soil) following freezing (values relative to bacteria concentration at the start of the experiment (data from Unc and Goss, 2006a)).

In surface waters, sunlight reduces the longevity of bacteria, whereas factors, such as turbidity that decrease the transmission of light tend to increase survival time (Aramini et al., 2000). *Campylobacter* survival rates in freshwater can be up to 4 months with survival greatest at 4 °C (Rollins and Colwell, 1986; Thomas et al., 1999). Survival time, however, is highly dependent upon strain type, previous growth conditions, water quality, and environmental conditions (Buswell et al., 1998, 1999).

4. Pathways between hazard locations and water resources

Pathways are the routes that can bring about the transfer of pathogens at a source to the receptor (watercourse). Identification of pathways are required to build process-based predictive tools or to design strategies to prevent the contamination of water sources. Overland flow is one of the principal mechanism by which faecal pathogens reach surface waters (Davies et al., 2004). Significant numbers of bacteria reach groundwater by infiltration through soil and any underlying rock strata (Joy et al., 1998).

Water is the primary agent determining the movement of contaminants. Where precipitation (rainfall or irrigation) exceeds infiltration rate, ponding can promote preferential flow. Other preferential flow routes that develop include cracking soils, root and earthworm channels (Goss et al., 1984). Freeze–thaw cycles may also result in fractures. The installation of tile drains also provides some continuous porosity between the soil surface and the drain. The soil pore characteristics determine the ability for rapid conduit between the field and the surface water body into which the tile drains discharge. For example, preferential flow through macropore allowed manure liquids to move into subsurface drains within an hour after application (Fleming and Bradshaw, 1991, 1992a, b).

Manure can affect the partitioning of water in the period immediately after land application, but the direction of the change depends on both the manure type and the soil type. In loamy and finer-textured soils, the application of dilute liquid manure can both encourage surface runoff and enhance preferential flow. Until solid manure has been incorporated, it acts as mulch and encourages infiltration rather than surface runoff (Unc and Goss, 2006b).

The overland flow pathway can be moderated by practices that reduce surface runoff from unconfined or partially confined pathogen sources. Distance and land slope between hazards and water source determine the pathway potential (Fraser et al., 1998; Stephenson and Street 1978). Vegetation, crop residues and soil clods can reduce significantly source water contamination by trapping bacteria (Collins and Rutherford, 2004), while bare soil increases overland flow and reduces re-deposition of pathogens (Davies et al., 2004). Vegetated buffer strips are most effective when they increase infiltration into the soil, and this also increases their efficiency of contaminant removal from surface runoff (Coyne et al., 1998).

The other hydraulic pathway along which pathogens move involves movement through the soil matrix or at least through the pore system. However, attenuation of pathogens through the processes of adsorption, filtration and absorption (Medema et al., 1997; Collins and Rutherford, 2004) is dependent on soil's physical and chemical properties and land management (Xin and Boll, 2003; McKay et al., 2002; Huysman and Verstraete, 1993; Smith et al., 1985). Pathogen movement is also affected by chemical and physical properties of the waste (Unc and Goss, 2004).

Most enteric pathogens reach the soil in the biosolid material that contained them. Bacterial retention and transport depend on the hydrophobic and hydrophilic interactions between the cell surface, soil mineral and organic surfaces and the soluble and suspended components in the soil solution. Factors influencing the effectiveness with which soils retain bacteria and viruses include cation concentrations, clays, soluble organic concentrations, pH, isoelectric point of the viruses, and general chemical composition of the soil. Investigations conducted by Unc (2002) on bacterial transport through soils following land application of liquid swine manure and sold beef manure indicate that initial retention of faecal bacteria in soils can be enhanced at high ionic strength of the suspending solution after land application of manure. Subsequent dilution of the soil solution by incoming rain or irrigation favours re-suspension of initially retained microbial cells. However, presence of biosolids colloidal matter cancelled some of the effects of the increased ionic strength, favouring particle transport through the vadose zone. Thus, despite the complexity of the interactions between bacterial cells soils and suspending solution, organic matter in the biosolids reduces the variability in the retention behaviour given by the intrinsic properties of charged particles (i.e. bacterial cells). Initially charged particles are therefore more likely to remain in suspension and penetrate deeper into the soil profile in the presence of suspended organic matter than they otherwise would.

5. Structure of the risk indicator

The risk of contaminating drinking water from a resource within an agricultural watershed depends on the number and size of pathogen sources, the existence of an active transport pathway, and any attenuation that occurs along the pathway (Fig. 3). The first component of an index of risk from pathogens is the identification, locating and sizing of each potential hazard. This requires the establishment of an inventory of confined and diffuse sources that exist on each farm in the watershed. Principally, this requires obtaining the locations and size of manure storages and OSWDS. Such information may be available from aerial photographs or satellite imagery, and the necessary validation of the interpretation is all the new material that would be necessary. Because of the influence of animal species, age, and management on the shedding of pathogens, the pathogen load in each hazard needs to be determined as a function of animal type, numbers and demography. In some jurisdictions, the required information on animals is available from census returns; in others it will need to be collected through questionnaires sent to producers. Basic information on animal and land management practices may well be obtained through the same mechanisms.

The second component is a compilation of information of die-off and growth rates for the different pathogen species in the different locations and media within potential pathways of movement. The boundaries between locations and media are potential barriers to further movement. The attenuation in pathogen numbers across these boundaries needs also to be compiled. In combination, these data sets permit the reduction in the strength of the threat over time to be calculated for each pathogen at each location in the farm environment as they move from the point of excretion along defined pathways and reach the different barriers in the farm system. The barriers that need to be identified along the pathogen migration pathway include the delay between excretion and entry into temporary and longer-term storage, length of storage and treatment, conditions after land application and barriers during transport by infiltration or overland flow from a manure source and from a septic system source. A survey of farm management practices would help quantify the factors that influence the duration of passage along identified pathways. A climatic assessment tool is needed to identify the likelihood of significant transport events.

The third component is the identification of potential pathways from each source to the key water resources. Surface pathways could be identified using an established digital elevation model of the catchment together with soil and geological maps. The outcomes are locations of surface flows that intersect with surface water courses as well as those that lead to receiving locations, which form recharge foci for groundwater. To determine which pathways are active requires the implementation of a microbial source tracking protocol (Goss and Dunfield, 2004). Essentially, this likely requires molecular techniques to establish that microbes in the target water resource are of the same type as those present in any of the principal hazards on a given farm. Ideally, the same organism should be identifiable at an intermediate point along the pathway.

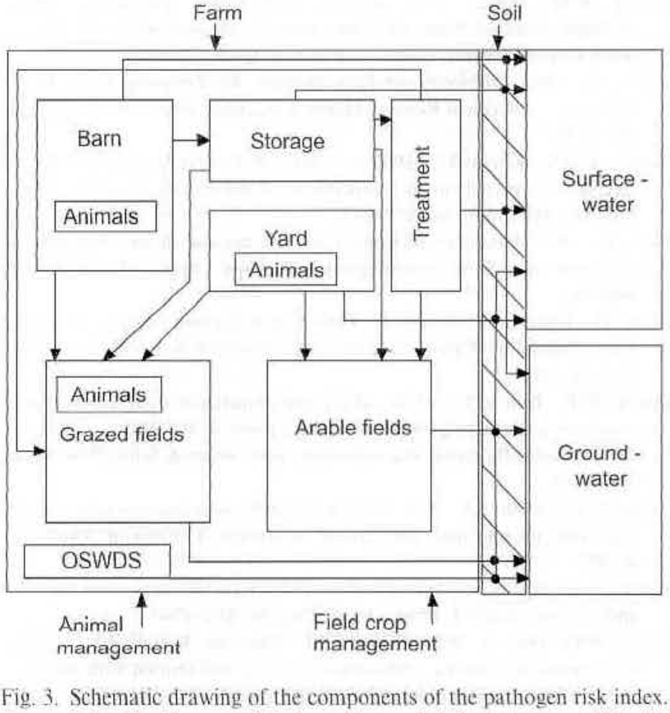


Fig. 3. Schematic drawing of the components of the pathogen risk index. Arrows below the framework indicate farmer decisions. Arrows within the framework indicate potential pathways for pathogen movement.

6. Conclusions

The development of a risk-based index of the potential for pathogens from agricultural activity to impact source waters is required as an interim stage in the establishment of a fully quantitative microbial risk assessment approach to source water protection. Based on the incidence of manure spills, unconfined sources are far more likely to pose threats to water resources, but the failure of confined sources can deliver very large loadings. There are significant limitations to identify and quantify the robustness of different barriers in attenuating the movement of pathogens. Particular emphasis needs to be put on the monitoring of subsurface drainage outfalls, as these can provide a direct link between preferential flow paths in the soil with surface waters.

References

Abu-Ashour, J., Joy, D.M., Lee, H., Whiteley, H.R., Zelin, S., 1994. Transport of microorganisms through soil. Water Air Soil Pollut. 75, 141–158.

Aramini, J., McLean, M., Wilson, J., Holt, J., Copes, R., Allen, B., Sears, W., 2000. Drinking Water Quality and Health Care Utilization for Gastrointestinal Illness in Greater Vancouver. Health Canada, Ottawa 〈www.hc-sc.gc.ca/ehp/ehd/catalogue/bch_pubs/vancouver_dwq.-hym〉.

Avery, S.M., Walters, L.D., Hutchinson, M.L., 2005. Fate of *Escherichia coli* O157 and detection of *stx* phage during fermentation of maize, an animal feed stuff. Lett. Appl. Microbiol. 40, 99–105.

Bach, S.J., Selinger, L.J., Stanford, K., McAllister, T.A., 2005a. Effect of supplementing corn- or barley-based feedlot diets with canola oil on faecal shedding of *Escherichia coli* O157:H7 by steers. J. Appl. Microbiol. 98, 464–475.

Bach, S.J., Stanford, K., McAllister, T.A., 2005b. Survival of *Escherichia coli* O157:H7 in feces from corn- and barley-fed steers. FEMS Microbiol. Lett. 252, 25–33.

Barker, J., Humphrey, T.J., Brown, M.W.R., 1999. Survival of *Escherichia coli* 0157 in a soil protozoan: implications for disease. FEMS Microbiol. Lett. 173, 291–295.

Barrington, S.F., Denis, J., Patini, N.K., 1991. Leakage from two manure tanks. Can. Agric. Eng. 34, 393–399.

Bogosian, G., Sammons, L.E., Morris, P.J., O'Neil, J.P., Heitkamp, M.A., Weber, D.B., 1996. Death of the *Escherichia coli* K-12 strain W3110 in soil and water. Appl. Environ. Microbiol. 62, 4114–4120.

Buswell, C.M., Herlihy, Y.M., Keevil, C.W., Marsh, P.D., Leach, S.A., 1999. Carbon Load in aquatic ecosystems affects the diversity and biomass of water biofilm consortia and the persistence of the pathogen *Campylobacter jejuni* within them. J. Appl. Microbiol. 85, S161–S167.

Buswell, C.M., Herlihy, Y.M., Lawrence, L.M., McGuiggan, J.T.M., Marsh, P.D., Keevil, C.W., Leach, S.A., 1998. Extended survival and persistence of *Cambylobacter* spp. in water and aquatic biofilms and their detection by immunoflurorescent-antibody and -rRNA staining. Appl. Environ. Microbiol. 64, 733–741.

Byrd, J.J., Xu, H.S., Colwell, R.R., 1991. Viable but non-culturable bacteria in drinking water. Appl. Environ. Microbiol. 57, 875–878.

CAST (Council for Agriculture Science and Technology). 1994. Food-borne Pathogens: Risk and Consequences. Task Force Report 122, 87pp.

Center for Disease Control and Prevention, 1998. Surveys of waterborne disease outbreaks compiled by the Center for Disease Control and Prevention from 1986 to 1998. In: Morbidity and Mortality Weekly Reports. CDC, Atlanta, USA.

Chandler, D.S., Farran, I., Craven, J.S., 1981. Persistence and distribution of pollution indicator bacteria on land used for disposal of piggery effluent. Appl. Environ. Microbiol. 42, 453–460.

Collins, R., Rutherford, K., 2004. Modelling bacterial water quality in streams draining pastoral land. Water Res. 38, 700–712.

Cools, D., Merckx, R., Vlassak, K., Verhaegen, J., 2001. Survival of *E. coli* and *Enterococcus spp.* derived from pig slurry in soils of different texture. Appl. Soil Ecol. 17, 53–62.

Coyne, M.S., Gilfillen, R.A., Villalba, A., Zhang, Z., Rhodes, R., Dunn, L., Blevins, R.L., 1998. Fecal bacteria trapping by grass filter strips during simulated rain. J. Soil Water Conserv. 53, 140–145.

Crane, S.R., Moore, J.A., Grismer, M.E., Miner, J.R., 1983. Bacterial pollution from agricultural sources: a review. Trans. A.S.A.E. 26, 858–872.

Davies, C.M., Long, J.A., Donald, M., Ashbolt, N.J., 1995. Survival of fecal microorganisms in marine and freshwater sediments. Appl. Environ. Microbiol. 61, 1888–1896.

Davies, C.M., Ferguson, C.M., Kaucner, C., Krogh, M., Altavilla, N., Deere, D.A., Ashbolt, N.J., 2004. Dispersion and transport of Cryptosporidium oocysts from fecal pats under simulated rainfall events. Appl. Environ. Microbiol. 70, 1151–1159.

Day, L., 2004. Septic systems a potential pollution source in the Cannonsville Reservoir Watershed, New York. J. Environ. Qual. 33, 1989–1996.

Dean, D.M., Foran, M.E., 1991. The Effect of Farm Liquid Waste Application on Receiving Water Quality. Final Report RAC Projects 430G and 512G. Ausable-Bayfield Conservation Authority, Exeter, Ontario.

Duffy, G., 2003. Verocytoxigenic *Escherichia coli* in animal feces, manures and slurries. Appl. Microbiol. 94, 94–103.

Duncan, I.J.H., Clark, E.A., Maitland, K., 1998. Livestock Behaviour in and near Watercourses in Ontario: 3 Year Summary. Unpublished report. Animal and Poultry Science and Plant Agriculture, University of Guelph, Guelph, Ontario.

Erlandsen, S.L., Mayer, E.A., 1984. *Giardia* and Giardiasis Biology, Pathogenesis and Epidemiology. Plenum Press, New York, 407pp.

Feachem, R.G., Bradley, D.J., Garelick, H., Mara, D.D., 1983. Sanitation and Disease: Health Aspects of Excreta and Wastewater Management. Wiley, Chichester, UK.

Fenlon, D.R., Ogden, I.D., Vinten, A., Svoboda, I., 2000. The fate of *Escherichia coli* and *E. coli* O157 in cattle slurry after application to land. Soc. Appl. Microbiol. 88, 149S–156S.

Feresu, S.B., Van Sickle, J., 1990. Coliforms as a measure of sewage contamination of the River Zambezi. J. Appl. Bacteriol. 68, 397–403.

Fleming, R., Fraser, H., 2000. Impacts of Winter Spreading of Manure on Water Quality: Literature Review. Ridgetown College, University of Guelph, Ridgetown, Ontario, Canada.

Fleming, R., McLelland, J., Alves, D., Alves, D., Hilborn, D., Pinter, K., Alpine, M., 1997. Crystosporidium in livestock manure storage and surface water in Ontario. Final report to Ontario Federation of Agriculture, Toronto, Ontario, Canada.

Fleming, R.J., Bradshaw, S.H., 1991. Macropore Flow of Liquid Manure. Paper No. 91-241, Canadian Society of Agricultural Engineering, Saskatoon, SK, Canada.

Fleming, R.J., Bradshaw, S.H., 1992a. Detection of soil macropores using smoke. Paper No. 92-103. Canadian Society of Agricultural Engineering, Saskatoon, SK, Canada.

Fleming, R.J., Bradshaw, S.H., 1992b. Contamination of Subsurface Drainage Systems during Manure Spreading. Paper No. 92-2618, A.S.A.E. St. Joseph, MI.

Foran, M.E., Dean, D.M., Taylor, H.E., 1993. The land application of liquid manure and its effect on tile drain water and groundwater quality. In: Agricultural Research to Protect Water Quality: Proceedings of the Conference. February 21–24. Minneapolis, MN. Soil and Water Conservation Society, Ankeny, IA, USA, pp. 279–281.

Fraser, R.H., Barten, P.K., Pinney, D.A., 1998. Predicting stream pathogen loading from livestock using a geographical information system-based delivery model. J. Environ. Qual. 27, 935–945.

Fuller, R., 1999. Probiotics for farm animals. In: Tannock, G.W. (Ed.), Probiotics: A Critical Review. Horizon Scientific Press, Andover, UK, pp. 15–21.

Gagliardi, J.V., Karns, J.S., 2000. Leaching of *Escherichia coli* 0157:H7 in diverse soils under various agricultural management practices. Appl. Environ. Microbiol. 66, 877–883.

Gale, P., 2005. Land application of treated sewage sludge: quantifying pathogen risk from consumption of crops. Appl. Microbiol. 98, 380–396.

Gary, H., Johnson, S., Ponce, S., 1983. Cattle grazing impact on surface water quality in a Colorado front range stream. J. Soil Water Conserv. 38, 124–128.

Gerba, C.P., Bitton, G., 1984. Microbial pollutants: their survival and transport pattern to groundwater. In: Gerba, C.P., Bitton, G. (Eds.), Groundwater Pollution Microbiology. John Wiley & Sons, New York, pp. 66–88.

Gerba, C.P., Smith, J.E., 2005. Sources of pathogenic microorganisms and their fate during land application of wastes. J. Environ. Qual. 34, 42–48.

Gerba, C.P., Wallis, C., Melnick, J.L., 1975. Fate of wastewater bacteria and viruses in soil. J. Irrig. Drain. Eng. 54, 157–174.

Gessel, P.D., Hansen, N.C., Goyal, S.M., Johnston, L.J., Webb, J., 2004. Persistence of zoonotic pathogens in surface soil treated with different rates of liquid pig manure. Appl. Soil Ecol. 25, 237–243.

Glass, E.J., 2004. Genetic variation and responses to vaccines. Anim. Health Res. Rev. 52, 197–208.

Goss, M.J., Dunfield, K.E., 2004. Tracking Sources of Microbial Impairment to Water Quality in Rural Ontario. Report on the Workshop held at the University of Guelph, Arboretum Centre, 5–6th April, 2004. Centre for Land and Water Stewardship, University of Guelph, Guelph, Ontario, 12pp.

Goss, M.J., Boone, F.R., Ehlers, W., White, I., Howse, K.R., 1984. Effects of soil management practice on soil physical conditions affecting root growth. J. Agric. Engng. Res. 30, 131–140.

Goss, M.J., Barry, D.A.J., Rudolph, D.L., 1998. Groundwater contamination in Ontario farm wells and its association with agriculture. 1. Results from drinking water wells. J. Contamin. Hydrol. 32, 267–293.

Gyles, C., 2000. *E. coli* O157:H7—Global perspective. In: Canadian Cattleman's Association (CCA) *E. Coli* O157:H7 Workshop. 27 and 28 November 2000, pp. 9–23.

Hagedorn, C., McCoy, E.L., Rahe, T.M., 1981. The Potential for Ground Water Contamination from Septic Effluents. J. Environ. Qual. 10, 1–8.

Hinton, M., Bale, M.J., 1991. Bacterial pathogens in domesticated animals and their environment. J. Appl. Bacteriol. 70, S81–S90.

HMSO, 1993. Interim report on Campylobacter. Advisory Committee on the Microbiological Safety of Food. HMSO, London, 107pp.

Hutchinson, M.L., Walters, L.D., Avery, S.M., Synge, B.A., Moore, A., 2004. Levels of zoonotic agents in British livestock manures. Lett. Appl. Microbiol. 39, 207–214.

Huysman, F., Verstraete, W., 1993. Effect of cell surface characteristics on the adhesion of bacteria to soil particles. Biol. Fertil. Soils 16, 21–26.

Jiang, X., Morgan, J., Doyle, M.P., 2002. Fate of *Escherichia coli* O157:H7 in manure-amended soil. Appl. Environ. Microbiol. 68, 2605–2609.

Joel, V., Karns, S.J., 2000. Leaching of *Escherichia coli* O157:H7 in diverse soils under various agricultural management practices. Appl. Environ. Microbiol. 66, 887–893.

Jones, K., Obiri-Danso, K., 1999. Non-compliance of beaches with the EU directives of bathing water quality: evidence of non-point sources of pollution in Morecambe Bay. J. Appl. Microbiol. 85, 101S–107S.

Joy, D., Lee, H., Reaume, C.M., Whitely, H.R., Zelin, S., 1998. Microbial contamination of subsurface tile drainage water from field application of liquid manure. Can. Agric. Eng. 40, 153–160.

Kearney, T.E., Larkin, M.J., Levett, P.N., 1993. The effect of slurry storage and anaerobic digestion on survival of pathogenic bacteria. J. Appl. Bacteriol. 74, 86–93.

King, D.J., Watson, G.C., Wall, G.J., Grant, B.A., 1994. The Effects of Livestock Manure Application and Management on Surface Water Quality. Summary Technical Report: GLWQP-AAFC Pest Management Research Centre, Agriculture and Agri-Food Canada London, Ontario, Canada.

Kudva, I.T., Blanch, K., Hovde, C.J., 1998. Analysis of *Escherichia coli* O157:H7 survival in ovine or bovine manure and manure slurry. Appl. Environ. Microbiol. 64, 3166–3174.

Lindenstruth, R.W., Ward, B.Q., 1948. Viability of *Vibrio fetus* in hay, soil and manure. J. Am. Vet. Med. Assoc. 113, 163.

MacLean, A.J., 1983. Pathogens of animals in manure: environmental impact and public health. In: Farm Animal Manures in the Canadian Environment. National Research Council of Canada Associate Committee on Scientific Criteria for Environmental Quality, Ottawa, Ontario, Canada, pp. 103–107.

Mawdsley, J.L., Badgett, R.D., Merry, R.J., Pain, B.F., Theodorou, M.K., 1995. Pathogens in livestock waste, their potential for movement through soil and environmental pollution. Appl. Soil Ecol. 2, 1–15.

McKay, D.L., Layton, A., Rietti-Shati, M., Driese, S., 2002. Transport of pathogen surrogates in saprolite subsoils in East Tennessee. 12th Tennessee Water Resources Symposium. Montgomery Bell State Park, TN, April 3–5, 2002, p. 6.

Medema, G.J., Bahar, M., Schets, F.M., 1997. Survival of Cryptosporidium parvum, *E. coli*, Faecal enterococci and *Clostridium perfringens* in river water–influence of temperature and autochthonous micro-organisms. Water Sci. Technol. 35, 249–252.

Monteith, H.D., Shannon, E.E., Derbyshire, J.B., 1996. The inactivation of bovine enterovirus and a bovine parovirus in cattle manure by anerobic digestion, heat treatment, gamma irradiation, ensilage and composting. J. Hygiene. 97, 175–184.

Mubiru, D.N., Coyne, M.S., Grove, J.H., 2000. Mortality of *Escherichia coli* O157:H7 in two soils with different physical and chemical properties. J. Environ. Qual. 29, 1821–1825.

Nicholson, F.A., Groves, S.J., Chambers, B.J., 2005. Pathogen survival during livestock manure storage and following land application. Bioresour. Technol. 96, 135–143.

Nodar, R., Acea, M.J., Carballas, T., 1992. Poultry slurry microbial population: composition and evolution during storage. Bioresour. Technol. 40, 29–34.

Nodar, R., Acea, M.J., Carballas, T., 1990. Microbial composition of poultry excreta. Biol. Wastes. 33, 95–105.

O'Connor, D.R., 2002. Part One: Report on the Walkerton Inquiry. The Events of May 2000 and Related Issues. Queen's Printer for Ontario, Ontario Ministry of the Attorney General, Toronto, Ontario.

Östling, C.E., Lindgren, S.E., 1991. Bacteria in manure and on manured and NPK fertilized silage crops. J. Sci. Food Agric. 55, 579–588.

Pell, A.N., 1997. Manure and microbes: public and animal health problem? J. Dairy Sci. 80, 2673–2681.

Peterson, T.C., Ward, R.C., 1989. Development of a bacterial transport model for coarse soils. Wat. Resour. Bull. 25, 349–357.

Plachá, I., Venglovsk, J., Sasáková, N., Svoboda, I.F., 2001. The effect of summer and winter seasons on the survival of *Salmonella typhimurium* and indicator micro-organisms during the storage of solid fraction of pig slurry. J. Appl. Microbiol. 91, 1036–1043.

Raina, P.S., Pollari, F.L., Teare, G.F., Goss, M.J., Barry, D.A.J., Wilson, J.B., 1999. The relationship between *E. coli* indicator bacteria in well-water and gastrointestinal illnesses in rural families. Can J. Publ. Health 90, 172–175.

Raya, R.R., Varey, P., Oot, R.A., Dyen, M.R., Callaway, T.R., Edrington, T.S., Kutter, E.M., Brabban, A.D., 2006. Isolation and characterization of a new T-even Bacteriophage, CEV1, and determination of its potential to reduce *Escherichia coli* O157:H7 levels in sheep. Appl. Environ. Microbiol. 72, 6405–6410.

Read, I.A., Svoboda, I.F., 1995. The effects of aerobic treatment on the survival of Cryptosporidium parvum oocysts in cattle slurry. In: Betts, W.B., Casemore, D., Frickers, C., Smith, H., Watkins, J. (Eds.), Protozoan Parasites and Water. Royal Society of Chemistry, Cambridge, UK, pp. 232–237.

Rollins, D.M., Colwell, R.R., 1986. Viable but nonculturable stage of *Campylobacter jejuni* and its role. Appl. Environ. Microbiol. 52, 531–538.

Roszak, D.B., Colwell, R.R., 1987. Survival strategies of bacteria in the natural environment. Microbiol. Rev. 51, 365–379.

Rowsell, J.G., Miller, M.H., Groenevelt, P.H., 1985. Self sealing of earthened liquid manure storage ponds: II. Rate and mechanism of sealing. J. Environ. Qual. 14, 539–543.

Rudolph, D.L., 2003. Performance of existing concrete manure storage facilities and association environmental risks. Report to Ontario Pork, University of Waterloo.

Rudolph, D.L., Barry, D.A.J., Goss, M.J., 1998. Contamination in Ontario farmstead domestic wells and its association with agriculture: 2. Results from multilevel monitoring well installations. J. Contam. Hydrol. 32, 295–311.

Russell, J.B., Diez-Gonzalez, F., Jarvis, G.N., 2000. Effects of diet shifts on *Escherichia coli* in cattle. J. Dairy Sci. 83, 863–873.

Sadovski, A.Y., Fattal, B., Goldberg, D., Katzenelson, E., Shuval, H.I., 1978. High levels of microbial contamination of vegetables irrigated with wastewater by the drop method. Appl. Environ. Microbiol. 36, 824–830.

Sinton, L.W., Finlay, R.K., Lynch, A.P., 1999. Sunlight inactivation of fecal bacteriophages and bacteria in sewage-polluted seawater. Appl. Environ. Microbiol. 65, 3605–3613.

Smith, M.S., Thomas, G.W., White, R.E., Ritonga, D., 1985. Transport of *Escherichia Coli* through intact and distributed soil columns. J. Environ. Qual. 14, 87–97.

Stephenson, G.R., Street, L.U., 1978. Bacterial variations in streams from a southwest Idaho rangeland watershed. J. Environ. Qual. 7, 150–155.

St. Jean, R., 1997. On-farm Manure Composting Techniques: Understanding Nitrogen and Carbon Conservation. Research Report 1.3. COESA Report No. RES/MAN-003/97. Prepared for Agriculture and Agri-Food Canada, London Research Centre, London, Ontario. Prepared by Ecologistics Limited, Waterloo, Ontario. ⟨http://res2.agr.gc.ca/london/env_prog/gp/gpres/report/rep13sum.html⟩.

Terzieva, S.I., McFeters, G.A., 1991. Survival and injury of *Escherichia coli*, Campylobacter jejuni and Yersinia enterocolitica in stream water. Can. J. Microbiol. 37, 785–790.

Thelin, R., Gifford, G.F., 1983. Fecal coliform release patterns from fecal material of cattle. J. Environ. Qual. 12, 57–63.

Thomas, C., Hill, D.J., Mabey, M., 1999. Evaluation of the effect of temperature and nutrients on the survival of *Campylobacter* spp. in water microcosms. J. Appl. Microbiol. 86, 1024–1032.

Unc, A., 2002. Importance of Manure Properties on Survival and Transport of Manure Bacteria in the Vadose Zone. PhD Dissertation, University of Guelph, Ontario, Canada.

Unc, A., Goss, M.J., 2004. Transport of bacteria from manure, and protection of water resources. Appl. Soil Ecol. 25, 1–18.

Unc, A., Goss, M.J., 2006a. Culturable *Escherichia coli* in soil mixed with two types of manure. Soil Sci. Soc. Amer. Proc. 70, 763–769.

Unc, A., Goss, M.J., 2006b. Impact of land application of organic wastes on soil hydraulic properties and on water partitioning. J. Environ. Eng. Sci. 5, 243–251.

Veira, D.M., Zimmerman, A.S., Corkal, D.R., 2003. Meeting the water requirements of a prairie livestock industry. Paper 9 Canadian Society of Animal Science Annual Meeting. June 10–13, 2003. University of Saskatchewan, Saskatchewan, Canada.

Wallis, P.M., Erlandsen, S.L., Issac-Renton, J.L., Olson, M.E., Robertson, W.J., Keulen Van, H., 1996. Prevalence of *Giardia* and *Cryptosporidium oocysts* and characteristics of *Giardia* spp. isolated from drinking water in Canada. Appl. Environ. Microbiol. 62, 2789–2797.

Wang, G., Zhao, T., Doyle, M.P., 1996. Fate of enterohemorrhagic *Escherichia coli* O157:H7 in bovine feces. Appl. Environ. Microbiol. 62, 2567–2570.

Whiteley, H., 1998. Effects of Cattle Access on Bacteria Concentrations in Streams. Unpublished report. School of Engineering, University of Guelph, Guelph, Ontario, Canada.

WHO, 2003. Emerging issues in water and infectious diseases. World Health Organization, Geneva, 24 pp.

Wyer, M.D., Kay, D., Dawson, H.M., 1996. Delivery of microbial indicator organisms to coastal waters from catchment sources. Wat. Sci. Tech. 33, 37–50.

Xin, D., Boll, J., 2003. Evaluation of attachment of *Cryptosporidium parvum* and *Giardia lambila* to soil particles. J. Environ. Qual. 32, 296–304.

Yates, M.V., 1985. Septic-tank density and groundwater contamination. Groundwater 23, 586–591.

Yeager, J.G., O'Brien, R.T., 1979. Enterovirus inactivation in soil. Appl. Environ. Microbiol. 38, 694–701.

ELSEVIER

Journal of Environmental Management 87 (2008) 633–638

Journal of
Environmental
Management

www.elsevier.com/locate/jenvman

A framework for valuing the health benefits of improved bathing water quality in the River Irvine catchment

E.K. Johnson[a], D. Moran[b], A.J.A. Vinten[c,*]

[a] *592 Quail Court Kingston, Ontario, Canada, K7M 8Z3*
[b] *Land Economy and Environment Group, SAC, West Mains Road, Edinburgh EH6 5AT, UK*
[c] *Macaulay Land Use Research Institute, Craigiebuckler, Aberdeen AB15 8QH, UK*

Received 30 April 2007; received in revised form 19 June 2007; accepted 19 June 2007
Available online 26 December 2007

Abstract

A simple model predicting bathing water concentrations of *Escherichia coli* from livestock in the Irvine catchment in SW Scotland has been adapted for intestinal enterococci (IE). This has been used to predict risk of bather illness by extrapolation of published data on bather IE exposure vs incidence of gastro-enteritis. Simulated reduction in the risk of illness by reduced faecal loading was multiplied by a willingness to pay for risk reduction to estimate the annual benefits of mitigation. Health benefits of reducing loading by 75% at Irvine Beach were estimated by a willingness to pay method to be about £276k pa. Estimated annualised costs of diffuse pollution mitigation measures across the catchment were higher (>£1 m), and it is very unlikely that 75% mitigation is achievable with current stocking rates. Further work should explore the influence of uncertainty of model parameters, and use emerging epidemiological information on specific zoonotic pathogens such as *E. coli* O157 and *Cryptosporidium*. Other components of the value of clean water should also be included to obtain a complete estimate of the cost:benefit of mitigation.

Keywords: Intestinal enterococci; Risk assessment; Bathing water; Valuation; Livestock

---

1. Introduction

High losses of faecal indicators to surface water make recreational bathing waters in SW Scotland vulnerable to microbial contamination (e.g. Kay et al., 2007; Vinten et al., 2005; Crowther et al., 2002). This is a significant public health issue (Lipp et al., 2001), and causes failure to meet EU Bathing Waters standards (EEC, 1976) which are designed to protect human health and ensure aesthetic standards for beaches. There are many ways to address the problem and society should adopt solutions that balance the costs and benefits. A large amount of current research is exploring applications of farm best management practices (BMPs) to reduce agriculture's contribution of faecal bacteria to coastal bathing waters (e.g. Cuttle et al., 2007; Dickson et al., 2005; Kay et al., 2007; Vinten et al., 2006; Haygarth, 2005). The EU Water Framework Directive (WFD) makes provision that if costs of mitigation are disproportionate, water quality mitigation can be derogated, at least temporarily. The provision of evidence of disproportionality can be done by assessing the relative costs and social benefits of pollution control. It is also important to recognise that ecological response to pollution control is highly non-linear, and aiming at inflexible thresholds may be inappropriate for achieving cost: effective environmental management (Statzner et al., 1997). Decisions on how to manage water quality could be based on private, social or ecological considerations depending on specific management goal(s) and the level of resourcing for implementation of improvement. In each case, classical economic theory suggests that resources should be deployed such that the marginal costs of pollution abatement equate marginal benefits of improvement at a relevant scale.

This paper presents a framework for valuing the health benefits to society of reducing faecal loading to bathing waters which may be achieved through BMP

---

[*] Corresponding author. Tel.: +44(0)1224498200.
E-mail address: a.vinten@macaulay.ac.uk (A.J.A. Vinten).

doi:10.1016/j.jenvman.2007.06.021

implementation. By relating a BMP-induced reduction in faecal loading to a decrease in risk of illness, benefits can be calculated using an estimate of society's willingness to pay for reduction in swimming-associated illness.

2. Model framework

Three components were involved in evaluating reduced faecal loading in the Irvine catchment in Ayrshire (Fig. 1). These will subsequently be linked to assessment of the cost effectiveness of mitigation.

2.1. Predicting bathing water quality

Vinten et al. (2005) devised a spreadsheet model to predict *Escherichia coli* in bathing water by accounting for inactivation, sedimentation, transport and mixing in the Irvine Beach/River catchment system for various loading in the catchment. Irvine Beach is located in Ayrshire in the west of Scotland (Grid Reference: NS305375). Irvine catchment has an area of 380 km^2. We have modified this model so IE levels in bathing waters could be predicted rather than *E. coli*, using changes to key parameter values based on literature, summarised in Table 1. Sinton et al. (1993), Reddy et al. (1981) and Geldreich (1978) all give values for numbers of IE relative to *E. coli* in cattle faeces of about 5 to 1. We have made use of literature to improve our estimate of the die-off of IE vs *E. coli*. Craig et al. (2001) provide *E. coli* and IE decay rates in water and sediment. The slope of the regression gives a half-life ratio of 2.75 ($p<0.001$, $\mathrm{SE}=0.50$, $n=20$, $r^2=0.62$). We have assumed that settling and transport properties are similar for the two indicators. The relative levels in animal faeces and inactivation factors show considerable variation in the literature, so these figures should only be considered indicative at this stage.

Bathing water quality predictions were made for current and 25–75% reduction in livestock loadings. Summer discharge percentiles were calculated using 14 years (1989–2002) of River Irvine data provided by E. Jow of Scottish Environmental Protection Agency (personal communication).

Fig. 1. Steps involved in determining the value of faecal loading reductions in the Irvine catchment.

Table 1
Correction factors used with model of Vinten et al. (2005) to predict IE concentrations in bathing water instead of *E. coli*

Parameter	Correction factor for IE/*E. coli*	References
Levels in animal faeces	5.6	Sinton et al. (1993), Reddy et al. (1981) and Geldreich (1978)
Half lives in soil and water	2.75	Craig et al. (2001)

2.2. Bathing water quality and risk of illness

A dose-response function between the concentration of IE in bathing water and the probability of contracting gastro-enteritis was determined by Kay et al. (1994) and Wyer et al. (1999), based on risk of contracting illness from bathing water quality determined by the World Health Organization using a risk function of bathing water quality throughout Europe (WHO, 2001, Fig. 4.3). This relationship has data for up to 39% probability of infection, and the authors suggest a maximum value of 39% be used to predict probability of infection above the data range. However, a Beta-Poisson relationship (e.g. Haas et al., 2000) is widely used to predict risk of illness, and we have used a least squares method to fit the data from WHO (2001) to a modified version of this curve with a zero response threshold:

$$r = 1 - [1 + \{((c - c_{\mathrm{NR}})/N_{50})(2^{1/\alpha} - 1)\}]^{-\alpha} \qquad (1)$$

where $r=$ excess probability of illness associated with bathing, $c=$ concentration of IE in bathing water (cfu/100 mL), $c_{\mathrm{NR}}=$ concentration of IE at which no excess probability of illness occurs, $N_{50}=$ dose for 50% infection rate, $\alpha=$ fitting constant.

The values of c_{NR}, α and N_{50} obtained were 31.8 cfu/100 mL, 1.0cfu/100 mL and 198 cfu/100 mL. This equation was linked to model simulations of IE in bathing water to give the risk of contracting gastro-enteritis as a function of faecal loading and river discharge (step 2 in Fig. 1), *including extrapolation where necessary beyond the observed data range.*

2.3. Economic valuation of reduced risk of illness

Public willingness to pay (WTP) for a reduction in risk of illness resulting from swimming in contaminated waters was determined using a benefit transfer from a stated preference study in England and Wales (EFTEC, 2002). The EFTEC study determined consumers' WTP for a reduction in risk of illness from swimming in contaminated bathing water. Valuations from the study indicate that respondents are willing to pay between £1.10 and £2.00 per household per year for a 1% reduction in the risk of suffering stomach upset (EFTEC, 2002) and we assumed a WTP of £1.10 per 1% reduction in risk. For comparison with Scottish bathing waters, a direct unadjusted mean value was used to determine the benefits of increased bathing water quality resulting from the reduction in faecal loading in the Irvine catchment. The EFTEC study is a suitable comparative site for Irvine Beach because of similarities listed in Table 2. The annual benefit accruing to Irvine Beach from reducing faecal loading was calculated by accounting for the likelihood of high discharge events.

Benefits were calculated at each percentile flow (above the 72nd) for a range of reductions in loading from 25% to 75%.

A spreadsheet model was developed to determine the benefits accruing to society from increased bathing water quality. Table 3 gives the equations used in the development of this model. Risk of illness percentages were calculated at various discharge percentiles for each loading reduction. The differences between risk of illness at full IE loading and the other loadings were determined. This accounts for the risk reduction expected by reducing IE loading.

Benefits accruing to the population were calculated for each discharge percentile. These values were aggregated across all percentile flows and corrected for the frequency of occurrence to determine total annual benefits (TB). Because the entire Scottish population was used, the total benefit value had to be corrected to elicit benefits occurring only at Irvine Beach. A 2004 study indicated that 3% of all Scottish beach visits were made to Irvine Beach (Scottish Executive Social Research, 2004). The total benefits to Scotland were therefore multiplied by 3% to calculate the benefits at Irvine Beach. Benefits accruing over a 25 year time horizon were discounted to their present value using a 6% discount rate. This rate maintains parity between values used in evaluating costs of agricultural infrastructure and machinery. Strictly speaking however, there are good arguments for treating cost and benefit streams differentially.

Using the above framework, benefits from reducing faecal loading were calculated for the River Irvine catchment.

Table 2
Comparison of site characteristics between proposed Scottish benefit transfer and EFTEC WTP study in England and Wales

	England and Wales	Scotland
Environmental quality parameter	Bathing water quality	Bathing water quality
Households	24,000,000	2,200,000
Population	52,900,000	5,062,000
Average household size	2.204	2.3
Mean weekly household income	£302	£296

3. Results

The simulated IE concentrations at Irvine beach are shown in Fig. 2. The simulations predict a compliance rate for the current guideline standard of 100 IE/100 mL for

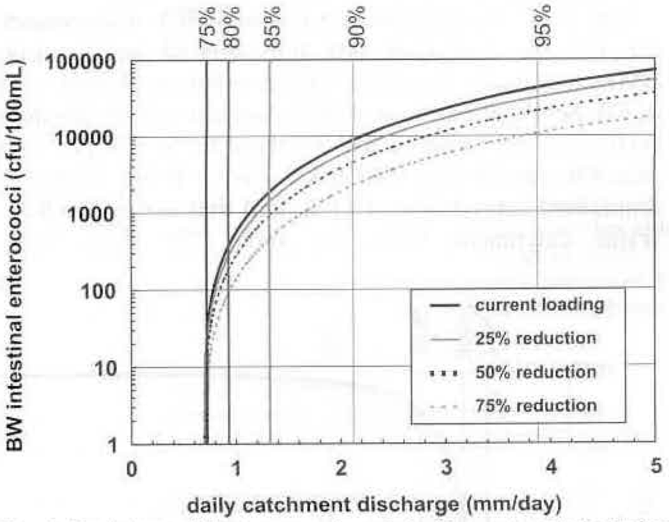


Fig. 2. Predictions of IE concentrations in bathing water at the Irvine Beach at various river Irvine scaled discharges. Vertical lines are River Irvine flow percentiles.

Table 3
Parameters used to model social benefits from increased bathing water quality

Parameter	Equation	Components
Risk of illness	$r = 1-[1+\{((c-c_{\mathrm{NR}})/N_{50})(2^{1/\alpha}-1)\}]^{-\alpha}$	• c = IE concentration in bathing water • c_{NR} = concentration of IE at which no excess probability of illness occurs • α, N_{50} = fitting constants.
Risk of illness differences	$\Delta r = r_{\mathrm{full}} - r_{\%\mathrm{reduction}}$	• r_{full} = risk of illness at full IE loading • $r_{\%\mathrm{reduction}}$ = risk of illness with reduction in loading
Social benefits at each discharge percentile	$B_f = \Delta r(\mathrm{WTP})h$	• WTP = household WTP to pay for risk reduction • h = number of households
Total annual benefits (TB)	$\mathrm{TB} = \sum_{f=0}^{f=100} \frac{\Delta r(\mathrm{WTP})h}{100}$	• f = percentile flow.
Lifetime benefits (LB)	$\mathrm{LB} = \sum_{t=0}^{t=25} \mathrm{TB}(1+i)^{-t}$	• TB = future value (in year t) • i = interest rate

75–80% of the time compared with the guideline compliance rate of 90% required by the regulations (SEERAD, 2001). Reduction in faecal loading to land by 75% only increases the compliance rate to about 80%. This assessment suggests that failure of the IE standard is likely to be more frequent than failure of the faecal coliform guideline standard, determined by Vinten et al. (2005).

The risk of illness was determined for current loadings and for 25%, 50% and 75% reduction in IE load at various levels of discharge (Fig. 3). Table 4 shows the annual benefit, as well as aggregated benefit over 25 years of improved bathing water quality at Irvine Beach, using a discount rate of 6% per year. Fig. 3 shows that risk of illness is insignificant at all IE input rates below the 75percentile flow. During 80-percentile flow ($4.1\,m^3/s$ or 0.93 mm/day over the $380\,km^2$ catchment), faecal loading needs to be reduced by $>75\%$ to give a risk of illness $<20\%$.

The WTP for avoidance of illness of BW improvement can now be compared with the costs of implementing BMPs to attempt to control diffuse pollution. Recent work in SW Scotland has estimated the annual cost of installing BMPs on two farms in SW Scotland to be £3k per farm plus 85k capital cost (SEERAD, 2003). If we assume an annualised capital cost of £17k, and that across the River Irvine catchment there are about 528 dairy farms (Merrilees, personal communication), total annualised mitigation costs are $(3+17)\times 528 = £1560k$. From Table 4, even with 75% reduction in faecal loading, annual benefits are only £276k. BMPs to assist in bathing water quality improvements include measures such as stock reduction, fencing, buffer strips, constructed wetlands, slurry storage, cattle access routes, and separating clean and dirty water. In practice achieving $>50\%$ efficacy may be very difficult, particularly considering that BMP performance during high intensity flow events is critical. Dickson et al. (2005) found a 40% reduction in high flow faecal indicator loadings where BMPs had been installed in the Brighouse catchment in SW Scotland. If the entire Irvine catchment installed these practices, bathing waters may still pose a high risk of illness following storm events.

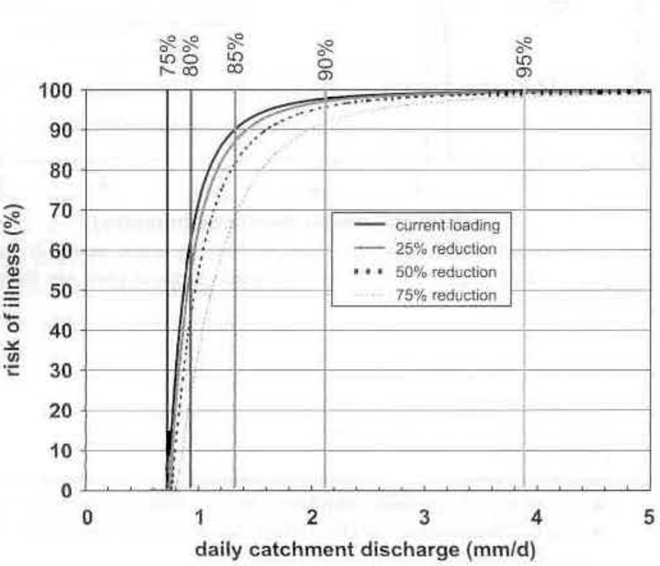


Fig. 3. Predicted risk of illness from bathing, in relation to discharge. Vertical lines are River Irvine flow percentiles.

Table 4
Expected lifetime benefits of reducing faecal loading in the Irvine catchment

Reduction in faecal loading in the Irvine catchment (%)	Annual benefit (in £k)	Aggregated benefits over 25 years at a 6% discount rate (in £k)
25	51	707
50	131	1802
75	276	3798

4. Discussion

These figures depend on the reliability of the extrapolation used in Eq. (1) for risk of illness, and the inactivation data for IE. Literature data on inactivation suggest longer survival of IE than *E. coli* in soil (Cools et al., 2001) but higher sensitivity to sunlight (Sinton et al., 2002). Noble et al. (2004) also give more rapid inactivation of IE than *E. coli* in freshwater and seawater, under low light intensity. There is also little information on the levels of IE in animal faeces. Improvement of the epidemiological database for such modelling should be a research priority.

We have assumed that reduction in stocking has a linear effect on faecal organism loads to water). Crowther et al. (2002) have shown a strong empirical relationship between faecal coliform (FC) counts in high flow discharges from rural catchments and the percentage of land which was either grazed or receiving manure in the catchment. However a simple linear relationship with stocking rates may not occur. Work examining the effects of destocking following foot and mouth disease in NW England has shown a much smaller reduction in faecal indicator loads than expected. This was partly because stream sediments act as long term reservoirs of faecal bacteria (Jamieson et al., 2005). Further work during the restocking process (2001–2002) shows significant correlations between geometric mean IE concentrations and cattle numbers in both winter ($r = 0.71$, $p<0.05$) and summer ($r = 0.94$, $p<0.001$). However the slope of the regressions between log transformed mean high flow IE concentrations and stocking density was 1.8–1.9, indicating a highly non-linear relationship between loads and stocking density, for the first 2 years of restocking (Sanders et al., 2005). In practice, this may mean that lowering stocking rates may have a stronger effect than we have assumed.

It is often assumed that areas impacted by diffuse, land-based faecal pollution may not present a health risk comparable to that produced by water impacted by human sewage effluents, due to the assumed lack of human pathogenic viruses in animal derived faecal matter. A study of the effects of diffuse pollution derived faecal bacteria by

Colford et al. (2007) found no correlation between water quality indicator levels for enterococcus, faecal coliform or total coliforms and the risk of illness. Although these authors did find a correlation between swimming and risk of diarrhoea, stomach cramps, skin rash and eye irritation this raises doubts about the illness risks attributable to bathing in waters affected by animal derived faecal indicators. However, the public health issue has been demonstrated with a number of outbreaks of *E. coli* O157 that have been clearly linked with recreational water contaminated by livestock (e.g. Ihekweazu et al., 2006). In order to clarify the situation, it is desirable to explore the risk of infection associated with specific pathogenic bacteria, protozoans and viruses, rather than use indicator bacteria. Zoonotic pathogens such as *E. coli* O157:H7, *Campylobacter* and *Cryptosporidium* spp. from farm livestock, birds, rodents etc contribute to the microbial pathogen loads to rivers. National and regional surveys of herd prevalence and incidence of EHEC *E. coli* O157 (e.g. Halliday et al., 2006; Hutchison et al., 2005) will enable estimates of risk from specific pathogens to be estimated.

Under the somewhat stringent assumptions set out in this paper, BMPs do not appear to offer a good return to outlay. However, we note a number of reasons that point to the benefits being underestimated. First is the treatment of health benefits through time and the use of the 6% discount rate for both costs and benefits. In reality we might want to draw on health economic arguments that suggest that the future value of health impacts should not be discounted at the same rate as the costs of infrastructure investments. More specifically, the latter are typically appraised in terms of the market rate of interest, which might be much higher than the time preference rates that society attaches to health improvements (see Krupnick, 2004). This argument can lead to rates used for health benefits of between 1% and 3%.

A second issue to consider is that BMP's can in many cases generate other non market benefits that have not been counted here. Significant biodiversity benefits for example can derive from some management options on farm (e.g. farm ponds and wetlands). Moreover, when faecal bacteria loads are reduced, nutrient loads to surface water may also be reduced, leading to a higher likelihood of achieving good water quality status. This paper has not attempted to value these benefits.

5. Conclusion

It is clear from these calculations that it is unlikely that the adoption of BMPs for bathing water protection can be justified on grounds of strict cost: benefit analysis for reduced health risk, unless the cost of fines for non-compliance with EU regulation are taken into account. However, since bathing and beach use are strongly linked to a perception of cleanliness, it may be that valuation based only on willingness to pay for avoidance of enteric illness may underestimate the true value of clean bathing water. Moreover, no assessments of the other benefits of cleaner water (in rivers, estuaries and sea) have been included in the valuation.

Further work should: (i) explore the influence of uncertainty of model parameters, especially with respect to epidemiology and sewage vs animal sources of pollution (ii) identify risks associated with specific zoonotic pathogens such as *E. coli* O157 and *Cryptosporidium*, and (iii) include other components of the value of clean water, before a complete and reliable picture of the cost: benefit of mitigation of faecal pollution can emerge.

Acknowledgement

We thank Scottish Executive Environment and Rural Affairs Department for funding this work under the Environment Research Program Work Package 3.5: Management to enhance Water Quality.

References

Colford Jr., J.M., Wade, T.J., Schiff, K.C., Wright, C.C., Griffith, J.F., Sandhu, S.K., Burns, S., Sobsey, M., Lovelace, G., Weisberg, S.B., 2007. Water quality indicators and the risk of illness at beaches with nonpoint sources of fecal contamination. Epidemiology 18, 27–35.

Cools, D., Merckx, R., Vlassak, K., Verhaegen, J., 2001. Survival of *E. coli* and enterococcus spp. derived from pig slurry in soils of different texture. Applied Soil Ecology 17, 3–62.

Craig, D.L., Fallowfield, H.J., Cromar, N.J., 2001. Comparison of decay rates of faecal indicator organisms in recereational coastal water and sediment. In: Proceedings of the second World Water Congress of the International Water Association, Berlin, Germany, 15–19 October 2001, 8 pp.

Crowther, J., Kay, D., Wyer, M.D., 2002. Faecal indicator concentrations in waters draining lowland catchments in the UK: relationships with land use and farming practices. Water Research 36, 1725–1734.

Cuttle, S.P., Macleod, C.J.A., Chadwick, D.R., Scholefield, D., Haygarth, P.M., Newell-Price, P., Harris, D., Shepherd, M.A., Chambers, B.J., Humphrey, R., 2007. An inventory of methods to control diffuse water pollution from agriculture. DEFRA Project ES02 03. Available from: 〈http://www.defra.gov.uk/farm/environment/water/csf/pdf/UserManual_Jan07.pdfE-Host〉 (accessed 18.06.07).

Dickson, J.W., Edwards, T., Jeffrey, W.A., Kay, D., 2005. Catchment scale appraisal of best management practices (BMPs) for the improvement of bathing water—Brighouse Bay. Report prepared for Scottish Environment Protection Agency (SEPA), Reference 230/4187.

Economics for the Environment Consultancy Ltd (EFTEC), 2002. Valuation of Benefits to England and Wales of Revised Bathing Water Quality Directive and other Beach Characteristics Using the Choice Experiment Methodology. Department for Environment, Food and Rural Affairs. Available from: 〈http://www.defra.gov.uk/environment/water/quality/bathing/bw_study4.htm〉 (accessed 16.08.05).

EEC (Commission of the European Communities), 1976. Directive 76/160/EEC of the European Parliament and of the Council concerning the quality of bathing waters. Official Journal of the European Communities L31, 1–7.

Geldreich, E.E., 1978. Bacterial populations and indicator concepts in faeces, sewage, stormwater and solid wastes. In: Berg, G. (Ed.), Indicator Viruses in Water and Food. Ann Arbor Science, MI, pp. 51–97.

Haas, C.N., Thayyar-Madabusia, A., Roseb, J.B., Gerbac, C.P., 2000. Development of a dose-response relationship for *Escherichia coli* O157:H7. International Journal of Food Microbiology 56, 153–159.

Halliday, J.E.B., Chase-Topping, M.E., Pearce, M.C., Mckendrick, I.J., Allison, L., Fenlon, D., Low, C., Mellor, D.J., Gunn, G.J., Woolhouse, E.J., 2006. Herd-level risk factors associated with the presence of phage type 21/28 *E. coli* O157 on Scottish cattle farms. BMC Microbiology 6, 99.

Haygarth, P., 2005. COST-DP: Cost-effective diffuse pollution mitigation. Report for Defra Project ES0121. Available from: 〈www.defra.gov.uk/science/default.htm〉.

Hutchison, M.L., Walters, L.D., Avery, S.M., Munro, F., Moore, A., 2005. Analyses of livestock production, waste storage, and pathogen levels and prevalences in farm manures. Applied and Environmental Microbiology 71, 1231–1236.

Ihekweazu, C., Barlow, M., Roberts, S., Christensen, H., Guttridge, B., Lewis, D., Paynter, S., 2006. Outbreak of *E. coli* O157 infection in the south west of the UK: risks from streams crossing seaside beaches. Euro Surveillance 11, 128–130.

Jamieson, R.C., Joy, D.M., Kostachuk, R., Gordon, R.J., 2005. Resuspension of sediment-associated *Escherichia coli* in a natural stream. Journal of Environmental Quality 34, 581–589.

Kay, D., Fleisher, J.M., Salmon, R.L., Jones, F., Wyer, M.D., Godfree, A.F., Zelenauch-Jacquotte, Z., Shore, R., 1994. Predicting likelihood of gastroenteritis from sea bathing: results from randomised exposure. Lancet 344, 905–909.

Kay, D., Aitken, M., Crowther, J., Dickson, I., Edwards, A.C., Francis, C., Hopkinds, M., Jeffrey, W., Kay, C., McDonald, A.T., McDonald, D., Stapleton, C.M., Watkins, J., Wilkinson, J., Wyer, M.D., 2007. Reducing fluxes of faecal indicator compliance parameters to bathing waters from diffuse agricultural sources: the Brighouse Bay study, Scotland. Environmental Pollution 147, 138–149.

Krupnick, A., 2004. Valuing Health Outcomes: Policy Choices and Technical Issues, Resources for the Future Report. Available from: 〈http://www.rff.org/Documents/RFF-RPT-ValuingHealthOutcomes.pdf〉 (accessed 28.05.07).

Lipp, E.K., Farrah, S.A., Rose, J.B., 2001. Assessment and impact of microbial fecal pollution and human enteric pathogens in a coastal community. Marine Pollution Bulletin 42, 286–293.

Noble, R.T., Lee, I.M., Schiff, K.C., 2004. Inactivation of indicator micro-organisms from various sources of faecal contamination in seawater and freshwater. Journal of Applied Microbiology 96, 464–472.

Reddy, K.R., Khaleel, R., Overcash, M.R., 1981. Behaviour and transport of microbial pathogens and indicator organisms in soils treated with organic wastes. Journal of Environmental Quality 10, 255–266.

Sanders, B., Anthony, S.G., Stapleton, C.M., Kay, D., Crowther, J., Wilson, J.G., 2005. The Impact of Destocking on the Microbiological Quality of Rivers in the Caldew Catchment–Vols. 1 & 2. Environment Agency Science Report SC020045/SR.

Scottish Executive Social Research, 2004. Bathing water use in Scotland: TNS Travel and Tourism. Scottish Executive Research. Available from: 〈http://www.scotland.gov.uk/Publications/2004/04/19210/35655〉 (accessed 22.04.07).

SEERAD, 2001. Bathing Waters (Classification) (Scotland) Regulations. Scottish Executive, Edinburgh.

SEERAD, 2003. Research and Design of Pilot Schemes to Minimise Livestock Pollution to the Water Environment in Scotland. Contract GLC 9/2. Final Report to the Scottish Executive. Scottish Executive Environment. Available from: 〈http://www.scotland.gov.uk/Topics/Environment/Water/15561/pollutionmainreport〉 (accessed 10.08.05).

Sinton, L.W., Donnison, A.M., Hastie, C.M., 1993. Faecal streptococci as faecal pollution indicators: a review. II. Sanitary significance, survival, and use. New Zealand Journal of Marine Freshwater Research 27, 117–137.

Sinton, L.W., Hall, C.H., Lynch, P.A., Davies-Colley, R.J., 2002. Sunlight inactivation of fecal indicator bacteria and bacteriophages from waste stabilisation pond effluent in fresh and saline waters. Applied and Environmental Microbiology 68, 1122–1131.

Statzner, B., Capra, H., Higler, L.W., Roux, A.L., 1997. Focusing environmental management budgets on non-linear system responses: potential for significant improvements to freshwater ecosystems. Freshwater Biology 37, 463–472.

Vinten, A.J.A., Lewis, D.R., McGechan, M., Duncan, A., Aitken, M., Hill, C., Crawford, C., 2005. Predicting the effect of livestock inputs of *E. coli* on microbiological compliance of bathing waters. Water Research 38, 3215–3224.

Vinten, A.J.A., Merrilees, D., Lewis, D.R., McGechan, M.B., Moran, D., Duncan, A., Aitken, M., Hill, C., Crawford, C., 2006. Implementation and evaluation of mitigation strategies for pollution control at a farm scale. SEERAD Report Number SAC/ 348/03.

World Health Organization (WHO), 2001. Bathing Water Quality and Human Health: Faecal Pollution, Geneva.

Wyer, M.D., Kay, D., Fleisher, J.M., Salmon, R.L., Jones, F., Godfree, A.F., Jackson, G., Rogers, A., 1999. An experimental health-related classification for marine waters. Water Research 33, 715–722.

ELSEVIER

Journal of Environmental Management 87 (2008) 639–648

Journal of
Environmental
Management

www.elsevier.com/locate/jenvman

Review

Network design for water quality monitoring of surface freshwaters: A review

Robert O. Strobl[a,*], Paul D. Robillard[b]

[a] *Water Resources Department, International Institute for Geo-Information Science and Earth Observation (ITC), Enschede, The Netherlands*
[b] *World Water Watch, Cambridge, MA, USA*

Received 3 May 2006; received in revised form 15 February 2007; accepted 11 March 2007

Abstract

To date, many water quality monitoring networks for surface freshwaters have been rather haphazardly designed without a consistent or logical design strategy. Moreover, design practices in recent years indicate a need for cost-effective and logistically adaptable network design approaches. There are many variables that need to be included in a comprehensive yet practical monitoring network: a holistic appraisal of the monitoring objectives, representative sampling locations, suitable sampling frequencies, water quality variable selection, and budgetary and logistical constraints are examples. In order to investigate the factors which affect the development of an effective water quality monitoring network design methodology, a review of past and current approaches is presented.

Keywords: Water quality; Monitoring network; Design methodology; Monitoring objectives; Sampling location; Sampling frequency; Surface freshwaters

---

Contents

---

1. Introduction

A variety of contaminants, in addition to a multitude of imprudent water management practices and destructive land uses, are currently threatening aquatic systems on a world-wide scale. In addition, it has been shown that water of good quality is a critical component for sustainable socio-economic development (Bartram and Balance, 1996). The impact and behavior of contaminants in an aquatic ecosystem are complex and may involve adsorption–desorption, precipitation–solubilization, filtration, biological uptake, excretion, and sedimentation–resuspension. Table 1 lists the important natural processes affecting water quality

---

[*] Corresponding author. Current address: European Commission, Joint Research Centre, IES, Ispra, Italy. Tel.: +39 0332 789391; fax: +39 332 785807.
E-mail address: robert.strobl@jrc.it (R.O. Strobl).

0301-4797/$ - see front matter © 2007 Elsevier Ltd. All rights reserved.
doi:10.1016/j.jenvman.2007.03.001

Table 1
Important natural processes affecting water quality (Bartram and Balance, 1996)

Process type	Major process within water body	Water body
Hydrological	Dilution	All water bodies
	Evaporation	Surface waters
	Percolation and leaching	Groundwater
	Suspension and settling	Surface waters
Physical	Gas exchange with atmosphere	Mostly rivers and lakes
	Volatilization	Mostly rivers and lakes
	Adsorption/desorption	All water bodies
	Heating and cooling	Mostly rivers and lakes
	Diffusion	Lakes and groundwater
Chemical	Photodegradation	Lakes and rivers
	Acid–base reactions	All water bodies
	Redox reactions	All water bodies
	Dissolution of particles	All water bodies
	Precipitation of minerals	All water bodies
	Ionic exchange	Groundwater
Biological	Primary production	Surface waters
	Microbial die-off and growth	All water bodies
	Decomposition of organic matter	Mostly rivers and lake
	Bioaccumulation	Mostly rivers and lakes
	Biomagnification	Mostly rivers and lakes

(Bartram and Balance, 1996). Besides natural processes affecting water quality, there are also anthropogenic impacts, such as man-induced point and non-point sources, xenobiotics, and alteration of water quality due to water use and river engineering projects (e.g., irrigation, damming, etc.) (Chapman, 1996).

The degradation of water resources has increased the need for determining the ambient status of water quality, in order to provide an indication of changes induced by anthropogenic activities. Water quality monitoring refers to the acquisition of quantitative and representative information on the physical, chemical, and biological characteristics of a water body over time and space (Sanders et al., 1983). In order to understand the process dynamics of a watershed, a well-designed water quality monitoring network identifies water quality problems while establishing baseline values for short- and long-term trend analysis. The need to evaluate observed water quality conditions and their suitability for the intended uses reflects a need for cost-effective and logistically practical water quality monitoring network design methods.

Starting in the 1960s and 1970s water quality monitoring has been developed to describe the general state of water quality. Typically, these early monitoring efforts involved arbitrary approaches without a consistent or logical design strategy. Sampling locations and frequencies were often determined by convenience or by other subjective criteria, and once the network was established, there was commonly no re-assessment of the effectiveness of the monitoring design (Harmancioglu et al., 1999; Tirsch and Male, 1984; Ward, 1996). In other words, a consistent and specific approach or methodology to water quality monitoring design was missing. The inadequacy of a proper network design methodology often results in water quality data collected with little analysis or ultimate purpose (GAO, 2000, 2002, 2004; NRC, 2002).

In recent years, the design of water quality monitoring networks has evolved to more focused topics, such as eutrophication, salinization, acidification, microbial, and heavy metal contamination among others. Where a specific contamination problem is known to exist (e.g., endocrine disruptors; Commission of the European Communities, 2004), ad hoc monitoring with selected variables or indicators is practiced. Furthermore, monitoring usually needs to be adapted for the aquatic system under study (river, headwater stream, lake, wetland, estuaries, reservoirs, etc.).

The importance of this topic has warranted a number of national and international conferences and workshops, such as the biannual National Monitoring conferences (held in the United States, 1998–2004) and the Monitoring Tailor-Made international workshops (held in the Netherlands, 1994–2003). These proceedings have tried to link monitoring at different scales in an interdisciplinary and integrative way. These scientific meetings conclude that monitoring is a helpful tool in early water pollution detection as well as in environmental remediation. The National Water Quality Monitoring Council has attempted to standardize monitoring data (e.g., Martin et al., 2001; Diamond et al., 2001).

Recent legislature is having an impact on monitoring network design. A prime example is the Water Framework Directive (WFD) of the European Community, which has set-up guidelines for comprehensive monitoring requirements for all river basins (WFD, 2003). These guidelines, of course, will affect the design of monitoring networks. New monitoring programs will include the following objectives (WFD, 2003):

- classify the status of all water bodies or groups of water bodies,
- support risk assessment procedures,
- design future monitoring programs,
- appraise long-term changes whose causes are both natural and anthropogenic,
- assess the compliance with objectives and standards,
- estimate the pollution load transfers across international boundaries or into seas,
- appraise the efficacy of measures applied to water bodies designated as at risk,
- ascertain formerly unidentified reasons for failure to achieve environmental objectives,
- assess the impact of accidental pollution,
- use in intercalibration exercises.

Therefore, the design of a monitoring network will include the type of water body, the classification system, the analysis of the associated pressures and risks, and the extent of the existing monitoring network. These pressures embrace the following for surface freshwaters (WFD, 2003):

- Rivers and lakes: nutrients, acidification, morphology, abstraction, and alien species.
- Additionally for rivers: hazardous substances, pesticides, mine waters, urban pressures, flow regulation, and pollution incidents.
- Transitional/coastal waters: nutrients, hazardous substances, organic enrichment, commercial fishing, morphology, and alien species.
- Additionally for transitional waters: abstraction, industrial intakes, and discharges.

Consequently, there are new challenges in network design. For surface waters the current status will be referenced to baseline conditions for each type of water body. An appraisal of the susceptibility of anthropogenic pressures will be quantified. Each appraisal will use available monitoring data. Existing data will vary greatly from country to country. The WFD seeks to harmonize the results of monitoring systems and ecological assessments rather than imposing a common ecological quality appraisal system in each country. Expert judgment as well as simulation modeling approaches can supplement data collection. The WFD approach includes flexibility in terms of monitoring frequencies and does not monitor all parameters in all cases. Thus, cost-effective and targeted monitoring programs can be designed. Hence, the WFD aims to improve the reliability of water body appraisals in terms of precision and confidence (WFD, 2003; Parr et al., 2002).

Nevertheless, the successful implementation of the WFD will rely on the availability of low-cost tools and technologies able to deliver appropriate and reliable data. An example of such tools are automatic monitoring stations, which have been cost-effectively employed already in numerous monitoring networks to capture the variability of contaminant concentrations or temporal changes in toxicity (Allan et al., 2006). Furthermore, in order to avoid the common practice of designing monitoring networks on subjective factors, attempts have been made to employ models for this purpose (e.g., Lo et al., 1996; Markus et al., 2003; Strobl et al., 2006a, 2006b). Lo et al. (1996) has analyzed the spatial variation of river water quality models to design a monitoring network for the Keelung River in northern Taiwan. Optimal sampling locations were selected using Kriging theory, whereas optimal sampling frequencies were determined by analysis of variance methods. Markus et al. (2003) used entropy and generalized least-square methods to appraise the regional value of stations in a monitoring network. Strobl et al. (2006a) have used a methodology that makes use of a geographical information system (GIS) and fuzzy logic approach, which was then converted into a functional model to determine critical sampling points (Strobl et al., 2006b). Further development of such models, be it of statistical or mechanistic nature, is still needed.

To appreciate the challenge of designing water quality networks, it is crucial to clearly define objectives and identify statistically acceptable assumptions. Assumptions are an inherent part of the monitoring network design process, mainly due to the stochastic influences on water quality variables in the aquatic environment. The number and type of simplifying assumptions made or allowed is dependent upon network objectives. Furthermore, assumptions in monitoring network design should be made relative to water quality hydrologic principles, applicable statistics, information utilization, and budget constraints (Sanders, 1980).

A logical and consistent design methodology should be developed that allows for more efficient and effective data collection and hence, more useful information extraction. Such an approach not only permits better water pollution control recommendations and better allocation of financial resources, but ultimately a better understanding of the ecosystems under study. The specific objective of this review of water quality monitoring network design for surface freshwaters is to establish which factors affect, and are important to monitoring network design. In addition, this review gives insight into past and current approaches to network design.

2. Monitoring network design

2.1. Monitoring and design considerations

Water quality monitoring is designed to capture data from which management and restoration information is extracted. It generally involves a vast number of activities, which are extensively described in Sanders et al. (1983) and

Table 2
Water quality monitoring activities (Sanders et al., 1983)

Main activity	Specific activities
1. Network design	Station locations Variable selection Sampling frequencies
2. Sample collection	Sampling techniques Field measurements Sample preservation Sampling points Sample transport
3. Laboratory analysis	Analysis techniques Operational procedures Quality control Data recording
4. Data handling	Data reception Screening and verification Storage and retrieval Reporting Dissemination
5. Data analysis	Basic summary statistics Regression analysis Water quality indices "Quality control" interpretation Time series analysis Water quality models
6. Information utilization	Information needs Reporting formats Operational procedures Utilization evaluation

summarized in Table 2. Network design comprises only a part of the total monitoring program. Ideally, water quality monitoring programs should adhere to generally accepted and standardized methodologies where applicable. As addressed above, such procedures normally include sampling protocol, sample preservation and handling, and laboratory analytical methods as well as field instrumentation. Complimentary to these procedures, quality assurance/quality control (QA/QC) should be included in a monitoring program. QA ensures that the final product meets the needs of its users and consists of an organized group of activities, whereas QC guarantees the compatibility of the results from different sampling sites as well as the reproducibility of data collected by different observers (Cline and Burkman, 1989).

Monitoring network design includes the number and spatial distribution of monitoring stations, sampling frequency, and parameter selection, as well as the temporal and spatial mode of data transfer. However, it is of utmost importance that the network objectives and accuracy criteria be defined as completely as possible (Steele, 1987).

Once monitoring objectives are defined, a methodology is needed to design the sampling network. However, a review of the literature suggests that there is no universally accepted methodology for the design of water quality monitoring networks. In fact, monitoring approaches being employed in watershed management are as varied as the watersheds being managed. As noted already by Sanders et al. (1983), the ability to choose adequate and rational monitoring network design procedures is seemingly as much an art as it is a science. This observation is as true today as it was two decades ago. Hence, there is an indisputable need to institute universally sound guidelines for water quality monitoring network design, or better yet, a design methodology that may be considered as acceptable and applicable to most common monitoring network design problems (Sanders, 1980). Only with clearly defined monitoring objectives and methodology can a network design produce an optimal configuration (Dawdy, 1979).

An optimal monitoring network design configuration satisfies monitoring objectives more efficiently and effectively than any other possible configuration. The water quality monitoring program must also be flexible, since the monitoring network design may need to be modified at a later stage, as deemed appropriate (Steele, 1987). The network design procedure is, therefore, an iterative process in which any information gained since the last iteration may lead to revisions in the network design at each new iteration. Without a monitoring feedback loop, in which the effects of those decisions are evaluated, the success as well as failure of previous management decisions cannot be accurately determined. However, this iterative process is commonly ignored once the water quality monitoring network has been established, leading often to the collection of unchanging and hence, useless and very expensive data. Most likely this "fear" of redesigning a water quality monitoring network is due to the belief that if existing programs are abandoned, the ability to determine trends may be forfeited. In addition, established water quality monitoring networks often include fixed sampling stations, which are only reluctantly deserted.

The representativeness of the collected samples needs to be considered. The US Geological Survey (1977) defines the term "representative" when the sample resembles the overall population in some way. To ensure that the data to be collected are suitable for comparative purposes, it is essential to standardize sampling bias in regard to sampling frequency, sampling location, sample collection, sample devices, sample handling and preservation, and sample identification (Canter, 1985).

Nevertheless, costs and availability of budgetary resources must also be considered when designing a monitoring network. Recognizing the economic constraints at an early stage will allow the designer to partition resources effectively (Sanders, 1980). As a result, the challenge in building an effective network design involves the collection of data, within the limits imposed by time and fiscal resources, which will render the maximum information for management purposes (Herricks, 1982). Furthermore, budgetary and logistical considerations almost always dictate that the data collection be taken discretely in time as well as in space. Another problem with respect to available financial resources in network design as investigated by Baumgart and Sperling (1984) is that, on the average, the ratio of measuring costs to costs for evaluation is about 5:1. In developing monitoring programs, the emphasis is usually placed on data collection and initial data analysis, instead of examining the reasons for monitoring or how the obtained data will be used as information in water quality management. Herricks (1982) notes that any monitoring program must be designed to fulfill existing as well as future information needs.

A review of monitoring approaches shows that many water quality monitoring network designs use a conceptual network as a guide in the design of the actual network (Woolfenden, 1984). A conceptual network represents an ideal network, in which the network is free from all budgetary constraints and, hence, where the network is solely based on hydrologic conditions, land use, existing water quantity and quality data, and the location of contamination sources. An ideal network can serve as a basis for future expansion of the actual network. For example, sampling stations can be added to the actual network if additional financial resources become available (Templin, 1984). Commonly, this conceptual network represents an aggregation of individual, single-purpose networks and needs to meet all of the individual objectives. A drawback of this procedure is, however, that specific, as well as extensive, knowledge and data of all the design parameters need to be available, which is commonly not the case. In fact, the purpose of the network is largely to obtain this knowledge of prevailing conditions and influences.

2.2. *Monitoring objectives*

In the initial stages of water quality monitoring network design, there is a need to identify the information expectations placed on the water quality monitoring network (Sanders et al., 1983). In most cases, the basic objective of a monitoring network is to supply the optimal level of information for water resources management and planning. However, before setting the monitoring objectives, the purposes of the network, in terms of the uses, and the users of the anticipated data need to be identified. It should be noted that data uses and users may vary temporally and spatially. In addition, potential future data needs should be identified and included into the network design. Once the purpose of the monitoring network has been established, an objective or set of objectives can be specified in terms of the desired information (WMO, 1994). Multiple objectives, multiple purposes, and multiple users need to be considered in a wide spectrum of contexts, including engineering, economic, social, and usually political. Sherwani and Moreau (1975) as well as Herricks (1982) have compactly summarized typical monitoring objectives.

Different objectives will consequently result in the design of different monitoring networks (Tirsch and Male, 1984). The monitoring objective generally determines the level of detail, the cost, and the necessary approach. Monitoring objectives need to be periodically checked to determine if they are being successfully fulfilled. If the monitoring objectives are found to be unmet, the current monitoring strategy must be modified or possibly completely altered in order to achieve the desired objectives. Furthermore, it is important to realize that a decision made without a clear understanding of its consequences often results in an outcome which will not satisfy management objectives. The periodic reviews of a water quality monitoring program should therefore, not only take into account changes in hydrologic conditions, but also shifts in management objectives (Kohonen, 1984; Herricks, 1982) as well as changes in technology and data analysis methods. For the case where there is more than one objective, priorities should be set for later evaluation. Priorities are in general not needed if all monitoring objectives can be met within the available budget (WMO, 1994). Therefore, the available budget not only influences the monitoring objectives, but also often limits network functions.

2.3. *Sampling location*

Sanders et al. (1983) consider the placement of a permanent sampling station probably the most critical design factor in a water quality monitoring network design. Nonetheless, in the past, criteria to establish station locations for representative sampling have received relatively little attention from governmental agencies responsible for water quality monitoring.

Many network designs subdivide the sampling locations into two groups, namely macrolocations and microlocations. Macrolocations are typically river reaches that are sampled within a watershed, whereas microlocations refer to sampling points that represent critical points, such as outfalls, determined relative to the macrolocation. Often in current water quality monitoring network design, macrolocations are allocated systematically, while microlocations are functions of macrolocations and critical locations (Harmancioglu et al., 1999).

In the past, water quality samples were often collected from points of mere convenience, such as bridges. Nevertheless, there also exist more structured approaches to designating sampling locations, such as Horton's (1945) method. This method defines stream order by specifying the smallest unbranched tributary in the headwaters of a watershed as first order, a stream consisting solely of first order tributaries as second order, etc. Sharp (1971) used Horton's approach for systematically designating sampling station locations by subdividing the stream network into equal portions of contributing tributaries. This procedure was developed to identify and isolate contamination sources. Sharp's (1971) procedure drew upon previous work by Shreve (1967), in which the pollution sources or the number of tributaries were used to identify sampling locations.

Other approaches included selecting sampling station sites as a function of possible stream standards violations or as a function of stream segments below outfalls (Beckers et al., 1972). Ward (1973), on the other hand, suggested locating sampling stations at critical quality points, such as each major source of pollution. Whichever approach is employed, a rational systematic approach is needed. In addition, the monitoring objectives need to be satisfied. Often, once fixed station locations have been installed, there is a tendency to limit further modifications of the network. However, fixed stations still can be "modified" in that sampling methods can be adjusted to account for any changes in physical, chemical, or biological conditions. Therefore, periodic reviews and analysis of the sampling program are essential.

2.4. Sampling frequency

Determination of sampling frequency is another critical, but often underemphasized area of concern in water quality monitoring network design. Sampling frequency primarily depends on the objective of the water quality monitoring network, but also on the relative importance of the sampling station location and the expected variability of the water quality data at each sampling station (Canter, 1985). Ideally, once the representativeness of water samples has been established in space through sampling location specification, then the sampling frequency should be defined so that the samples are also representative in time. Unfortunately, in the past, infrequent sampling (commonly monthly) has often been the standard procedure in the operation of fixed station monitoring networks. In addition, the sampling frequency was often determined by the available capacity of the analytical laboratory and the samples were frequently equally divided among the existing monitoring stations (WMO, 1994).

Sampling frequency should be adjusted so that water quality information can be gathered with a minimum of sampling effort. Although automatic monitoring has been applied in some network designs, it should be noted that a large portion of the costs for operating a monitoring network as well as for data management and QA/QC (including meta data) is directly related to the frequency of sampling. Limited financial resources most likely will not allow the same sampling frequency of all parameters at every sampling site.

By increasing the number of samples collected, a reduction in the standard error of the mean value can be realized. Since the standard error of the mean is inversely proportional to the square root of the number of samples, an increase in the number of observations will consequently only lead to a rather small overall gain in results. The greater the variation of the water quality, the greater the number of samples needed to obtain a statistically sound estimate that describes parameter behavior. Typically the mean is the most reported statistical parameter. A sampling frequency is selected that gives an estimate of the mean within a given confidence limit. The confidence limits of the mean quantify the choice of sampling frequency by relating sampling frequency to the water quality variation (Sanders et al., 1983). It should nevertheless be noted that, in practice, a water quality sampling program commonly consists of more than a single water quality variable and of more than one sampling location. Generally, it is desirable to extract equal information from each location, unless there is some way of prioritizing the sampling stations. Furthermore, if the means are approximately equal, then the confidence intervals about the means from one station to the next are equal. It is normally also desirable to have small confidence interval widths. One way to achieve this is by a proportional sampling procedure, in which the following equation is used to assign the number of samples to be collected at each station, n_i:

$$n_i = w_i N, \quad (1)$$

where w_i is the weighting factor for station i, where $\Sigma_{i=1}^{N} w_i = 1.0$, and N is the total number of samples in the water quality monitoring network. The weights in this equation are typically assigned as a function of population density, historical variance, mean variable values, and flow (Ward, 1978).

Little quantitative criteria specifying suitable sampling frequencies have been developed for the design of water quality monitoring networks. One approach used in the past assumed that the water quality variable concentrations are random, independent, and identically distributed. Using this approach, the number of samples per year is determined as a function of a specified confidence interval of the mean annual concentration:

$$N = [(t_{\mathrm{alpha}/2} S)/R]^{**}2, \quad (2)$$

where N is the number of equally (temporally) spaced samples collected per year; $t_{\mathrm{alpha}/2}$ is a constant which is a function of the level of significance, and the number of samples; S is the standard deviation of the water quality concentrations; and R is the specified half-width of the confidence interval of the annual mean. The above approach for specifying the sampling frequency, however, obviously depends on the chosen confidence level (e.g., 95%) as well as on the confidence interval width of the annual mean (Sanders et al., 1983).

More statistically advanced approaches involving fewer assumptions have been developed, but often rely on extensive time series data for determining the optimal sampling frequency. Unfortunately, with the general exception of mean daily discharge, water quality databases of adequate size, length, as well as reliability are commonly the limiting factor in applying these techniques.

2.5. Water quality variables

Both sampling location and sampling frequency are influenced by the water quality variable being monitored and, therefore, the selection of the specific variables of concern or interest is intrinsic to the design and subsequent operation of a water quality monitoring network. The monitoring network objectives or a precisely as possible defined information "need" is necessary in the appropriate selection of water quality variables. Careful and effective selection of water quality variables is necessary since determination of certain variables can be extremely expensive, either in terms of analytical laboratory costs or in specialized sampling or preservation techniques. Consideration should be given to reducing the number of parameters sampled without substantial loss of information. Fewer parameters would be easier to analyze and establish dependencies or correlations between various water quality variables, saving time and effort. In addition, such an analysis might reveal which variables could be neglected under a given set of circumstances. The water quality network would not just be collecting data, but would allocate financial resources to target state-controlling variables and develop recommendations to improve the physical, chemical, and biological integrity of the impacted water resources (Steele, 1987; Sanders et al., 1983; Canter, 1985).

The variables that characterize water quality may be classified in several ways, including their physical, chemical, and biological properties. Furthermore, the importance attached to each parameter is critical. This will vary with the type of water body in question, intended uses, and the objectives of the monitoring program. Physical variables of interest normally encompass discharge (for streams), water temperature, turbidity, specific conductance, particle size distribution, suspended sediment concentration, taste, and odor. Chemical water quality variables are commonly classified in terms of inorganic and organic categories. Basic variables commonly included in a water monitoring effort are listed in Table 3 (WMO, 1994). It should be noted that in recent years, besides the basic variables listed in Table 3, many monitoring programs have also included various environmentally detrimental variables, such as micropollutants and indicators of microbial contamination.

Table 3
Common water quality variables (WMO, 1994)

	Rivers	Lakes and reservoirs	Groundwater
General water quality			
Water discharge/level	x	x	x
Total suspended solids	x	—	—
Temperature	x	x	x
pH	x	x	x
Electrical conductivity	x	x	x
Dissolved oxygen	x	x	x
Transparency	—	x	—
Dissolved salts			
Calcium	x	x	x
Magnesium	x	x	x
Sodium	x	x	x
Potassium	x	x	x
Chloride	x	x	x
Fluoride	—	—	x
Sulfate	x	x	x
Alkalinity	x	x	x
Nutrients			
Nitrate plus nitrite	x	x	x
Ammonia	x	x	x
Total phosphorus, dissolved	x	x	—
Total phosphorus, particulate	x	x	x
Total phosphorus, unfiltered	x	x	—
Silica reactive	x	x	—
Organic matter			
Chlorophyll *a*	x	x	—

It is noteworthy that even though water quality variable classifications according to independently assumed variables are quite convenient and useful, the interplay between different variables needs to be considered as well. Landwehr (1974) noted that in order to increase the economic efficiency in operating a water quality monitoring network, detection and assessment of such interactions aid in determining which variables might potentially serve as indicators of a wider category of variables. This is especially desirable in light of the fact that there are hundreds of different environmentally hazardous chemicals present in ecosystems.

Even though it is not always practical to measure all water quality variables of interest or concern in a sampling effort, presently available alternatives to evaluate water quality information include statistical analysis and water quality modeling techniques. Several statistical techniques for transforming data into information and for indirectly supplying water quality values were given by Sanders et al. (1983). Furthermore, Sanders et al. (1983) note that a broad understanding of the interactions between water quantity and water quality variables is useful in solving various water quality related problems. Also, an understanding of the "natural" (i.e., pristine or pre-development) and man-affected conditions (i.e., post-development) is essential.

Special attention should also be paid to the importance of biological monitoring when setting up a monitoring network, as it is a rapidly growing field that nowadays incorporates ecological indicators (e.g., fish, benthos, diatoms, etc.), biological tests, chemical analyses of tissues, and exposed (caged) organisms. Biological analysis of an aquatic ecosystem generally reflects a much longer period of water quality conditions than traditional physical-chemical water quality monitoring. In many cases, biological evaluations regarding water quality have even detected mild, intermittent pollution, very often overlooked or not detected by standard chemical methods. Nevertheless, it should be noted that the trend of water quality improvement is typically first apparent in physio-chemical analyses and only with some delay in biological analyses since living organisms usually require time to adapt to new conditions. In contrast, the trend of quality deterioration is normally first evident by biological analyses and only much later by physiochemical analyses. Therefore, biological monitoring of water quality apparently has a predictive and signaling as well as a controlling function (Karr and Chu, 1998; Ometo et al., 2000).

The approaches for analyzing and interpreting biological communities relative to water quality assessment tend to be ad hoc and highly subjective. Therefore, in order to take advantage of the full potential locked into biological monitoring methods, reliable data interpretation and classification systems need to be developed. Most biomonitoring techniques will only provide reliable results if the investigator performing the field biological monitoring has a basic background in biology. On the other hand, routine biological monitoring is relatively inexpensive. It has also been noted that biological communities may be the only practical means of water quality assessment where criteria for specific ambient impacts do not exist, particularly in reference to non-point source pollution. Another advantage of biological monitoring is that it integrates the effects of mixed pollutants. On the other hand, since biological monitoring methods can only indirectly detect causes for a given change, chemical and physical monitoring will always remain an integral part of a meaningful monitoring activity (Dates and Byrne, 1997; Sponseller et al., 2001).

3. Conclusions

This review has given insight into the past and current strategies of water quality monitoring network design for surface freshwaters, along with their weaknesses and shortcomings. There is a plethora of considerations as well as issues that need to be addressed when designing an efficient and effective water quality monitoring network. Network design ideally needs to include a combined appraisal of the monitoring objectives, representative sampling locations, sampling frequencies, water quality variable selection, and budgetary and logistical constraints. To accomplish this seemingly overwhelming task, a methodology for water quality monitoring network design would be of utmost importance to ensure a valid, efficient, and lastly cost-effective design. This is particularly important since the costs of water quality monitoring networks can be extremely high, given specific information and management needs.

Another important aspect that this review has revealed is that in the past, the majority of water quality monitoring programs have been designed on a rather arbitrary basis. Even though much research has been undertaken since 1970 to improve water quality management decisions, much of it has not been implemented in present-day water quality monitoring programs since most of the research has been either too general, too specific (i.e., too case-limited), or simply difficult for a watershed manager to easily incorporate into a water quality monitoring network design, given time and budget constraints. World-wide attention is thrusting water quality monitoring efforts in a new and integrated direction. Hence, it is obvious that a practical, cost-effective water quality monitoring network design methodology, which allows a watershed manager to practically design a water quality monitoring network through a user-friendly interface, is needed. In addition, the design of an optimal (i.e., cost-effective) water quality monitoring network configuration must be able to identify the average and extreme water quality conditions, as well as any significant changes or trends.

The emergence of new technologies and analysis techniques need to be explored with respect to monitoring network design. GIS, remote sensing techniques, as well as artificial intelligence technologies induce innovation and new problem-solving criteria. In fact, Strobl et al. (2006a) have illustrated a methodology that uses these kinds of techniques for the design of water quality monitoring networks in smaller, even remotely located, watersheds with respect to total phosphorous monitoring.

This review leads to the following recommendations for the development of a methodology for designing monitoring networks for surface freshwaters:

1. A standardized monitoring network design methodology is essential to ensure an ecosystem-based management approach that considers human activities, their benefits, and their potential impacts within the context of the broader biological and physical environment.
2. The design methodology should establish what, where, when, and how to monitor, including definition of a set of core variables.
3. The design of a monitoring network needs to be periodically re-assessed and accordingly modified due to changing environmental conditions. This includes the periodic review of the design with respect to the measurement of variables that reflect the health of an ecosystem.
4. Successful monitoring should target issues that policy makers, watershed managers, scientists, and the public consider important. Input from these sectors should therefore be considered in the design of a network.
5. If the monitoring activities are to be used on a wider, say national, basis, designers should concur on a set of core variables to be measured at every sampling location, with flexibility for stakeholders to measure additional variables if desired or needed.
6. The design of the network should account for both effects-based monitoring, which measures the present condition of the water environment, as well as stressor-oriented monitoring, which measures parameters that are suspected or known to be associated with a negative impact on water environmental health.
7. The design should ideally include standardized laboratory procedures and techniques as well as universal QA/QC recommendations.
8. Data management protocols should be established and uniform data storage formats specified so that data can be broadly disseminated, accessed, and understood.

9. An efficient monitoring network design should not only be able to successfully track specific substances, but also be effective in helping understand how various ecosystem components interact and change over the long-term.

References

Allan, I.J., Vrana, B., Greenwood, R., Mills, G.A., Roig, B., Gonzalez, C., 2006. A "toolbox" for biological and chemical monitoring requirements for the European Union's Water Framework Directive. Talanta 69, 302–322.

Bartram, J., Balance, R., 1996. Water Quality Monitoring: A Practical Guide to the Design and Implementation of Freshwater Quality Studies and Monitoring Programmes. Chapman & Hall, London.

Baumgart, H., Sperling, F., 1984. Monitoring stations and water quality measurements on the river Lippe: equipment–results–costs. Water Science and Technology 16, 327–346.

Beckers, C.V., Chamberlain, S.G., Grimsrud, G.P., 1972. Quantitative methods for preliminary design of water quality surveillance systems. Socioeconomic Environmental Studies Series Report No. EPA-R5-72-001, US EPA, Washington, DC.

Canter, L.W., 1985. River Water Quality Monitoring. Lewis Publishers, Inc., Chelsea, MI.

Chapman, D. (Ed.), 1996. Water Quality Assessments. A Guide to the Use of Biota, Sediments and Water in Environmental Monitoring. Chapman & Hall, London.

Cline, S.P., Burkman, W.G., 1989. The role of quality assurance in ecological programs. In: Bucher, J.B., Bucher-Wallin, I. (Eds.), Air Pollution and Forest Decline. 14th International Meeting for Specialists on Air Pollution Effects on Forest Ecosystems, IUFRO P2.05, Interlaken, Switzerland, 2–8 October 1988, pp. 361–365.

Commission of the European Communities, 2004. Community strategy for endocrine disruptors—a range of substances suspected of interfering with the hormone systems of humans and wildlife. SEC(2004) 1372, Brussles, EC.

Dates, G., Byrne, J., 1997. Living Waters: Using Benthic Macroinvertebrates and Habitat to Assess Your River's Health. River Watch Network, Montpelier, VT, USA.

Dawdy, D.R., 1979. The worth of hydrologic data. Water Resources Research 15, 1726–1732.

Diamond, J., Eaton, A., Annis, C., Brass, H., Keith, L., Strong, A., McChesney, D., Shockey, M., 2001. Towards a definition of performance-based laboratory methods. National Water Quality Monitoring Council Technical Report 01-02, NWQMC, Washington, DC.

General Accounting Office, 2000. Water quality: Key EPA and State decisions limited by inconsistent and incomplete data. GAO Report Number GAO/RCED-00-54, GAO, Washington, DC.

General Accounting Office, 2002. Inconsistent State approaches complicate Nation's efforts to identify its most polluted waters. GAO Report Number GAO-02-186, GAO, Washington, DC.

General Accounting Office, 2004. Watershed management: better coordination of data collection efforts needed to support key decisions. GAO Report Number GAO-04-382, GAO Washington, DC.

Harmancioglu, N.B., Fistikoglu, O., Ozkul, S.D., Singh, V.P., Alpaslan, M.N., 1999. Water Quality Monitoring Network Design. Kluwer Academic Publishers, Dordrecht, The Netherlands.

Herricks, E.E., 1982. Biological monitoring: receiving system methodology based on community structure. Water Research 16, 141–153.

Horton, R.E., 1945. Erosional development of streams. Geological Society American Bulletin 56, 281–283.

Karr, J.R., Chu, E.W., 1998. Restoring Life in Running Waters: Better Biological Monitoring. Island Press, Washington, DC.

Kohonen, T., 1984. Automatic monitoring of river water quality. Water Science and Technology 16, 289–294.

Landwehr, J.M., 1974. Water quality indices—construction and analysis. Ph.D. Dissertation, University of Michigan, University of Microfilm no. 75-10, 212, University of Michigan, Ann Arbor, MI.

Lo, S.L., Kuo, J.T., Wang, S.M., 1996. Water quality monitoring network design of Keelung river, northern Taiwan. Water Science and Technology 34, 49–57.

Markus, M., Knapp, H.V., Tasker, G.D., 2003. Entropy and generalized least square methods in assessment of the regional value of streamgages. Journal of Hydrology 283, 107–121.

Martin, L., Ward, R.C., McBride, G.B., Loftis, J.C., 2001. Data analysis considerations in producing 'comparable' information for water quality management purposes. Technical Report No. 01-01, National Water Quality Monitoring Council, USGS, Reston, Virginia.

National Research Council, 2002. Opportunities to improve the US Geological Survey National Water Quality Assessment Program. National Academic Press, Washington, DC.

Ometo, J.P.H.B., Martinelli, L.A., Ballester, M.V., Gessner, A., Krusche, A.V., Victoria, R.L., Williams, M., 2000. Effects of land use on water chemistry and macroinvertebrates in two streams of the Piracicaba River basin, southeast Brazil. Freshwater Biology 44, 327–337.

Parr, T.W., Ferretti, M., Simpson, I.C., Forsius, M., Kovács-Láng, E., 2002. Towards a long-term integrated monitoring programme in Europe: network design in theory and practice. Environmental Monitoring and Assessment 78, 253–290.

Sanders, T.G., 1980. Water quality as a stochastic process. In: Proceedings of the ASCE National Convention, Portland, OR, April 14–18.

Sanders, T.G., Ward, R.C., Loftis, J.C., Steele, T.D., Adrian, D.D., Yevjevich, V., 1983. Design of Networks for Monitoring Water Quality. Water Resources Publications LLC, Highlands Ranch, CO.

Sharp, W.E., 1971. A topologically optimum water-sampling plan for rivers or streams. Water Resources Research 7, 1641–1646.

Sherwani, J.K., Moreau, D.H., 1975. Strategies for water quality monitoring. Water Resources Institute of the University of North Carolina, Report No. 107, University of North Carolina, Chapel Hill, NC.

Shreve, R.L., 1967. Infinite topologically random channel networks. Journal of Geology 75, 178–186.

Sponseller, R.A., Benfield, E.F., Valett, H.M., 2001. Relationships between land use, spatial scale and stream macroinvertebrate communities. Freshwater Biology 46, 1409–1424.

Steele, T.D., 1987. Water quality monitoring strategies. Hydrological Sciences Journal 32, 207–213.

Strobl, R.O., Robillard, P.D., Shannon, R.D., Day, R.L., McDonnell, A.J., 2006a. A water quality monitoring network design methodology for the selection of critical sampling points: Part I. Environmental Monitoring and Assessment 112, 137–158.

Strobl, R.O., Robillard, P.D., Day, R.L., Shannon, R.D., McDonnell, A.J., 2006b. A water quality monitoring network design methodology for the selection of critical sampling points: Part II. Environmental Monitoring and Assessment 122, 319–334.

Templin, W.E., 1984. Ground-water-quality monitoring network design for the San Joaquin Valley ground-water basin, California. USGS Water-Resources Investigations Report 83-4080, USGS, Denver, CO.

Tirsch, F.S., Male, J.W., 1984. River basin water quality monitoring network design: options for reaching water quality goals. In: Schad, T.M. (Ed.), Proceedings of the Twentieth Annual Conference of American Water Resources Associations. AWRA Publications.

US Geological Survey, 1977. Elements needed in design of a ground-water-quality monitoring network in the Hawaiian Islands. Water-Supply Paper W 2041.

Ward, R.C., 1973. Data acquisition systems in water quality management. Socioeconomic Environmental Series Studies Report No. EPA-R5-73-014, US EPA, Washington, DC.

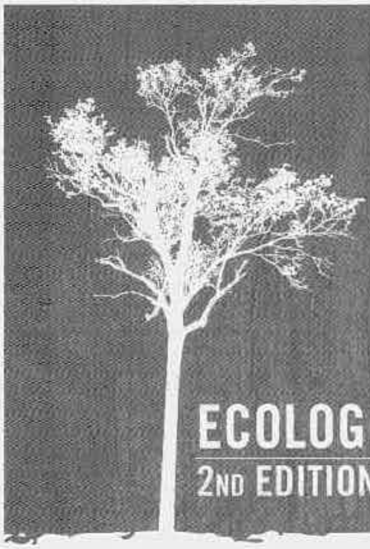
Ward, R.C., 1978. Evaluating the sampling frequencies of water quality monitoring networks. EP 1.23/8:600/7-78-169, US EPA, Las Vegas, NV.

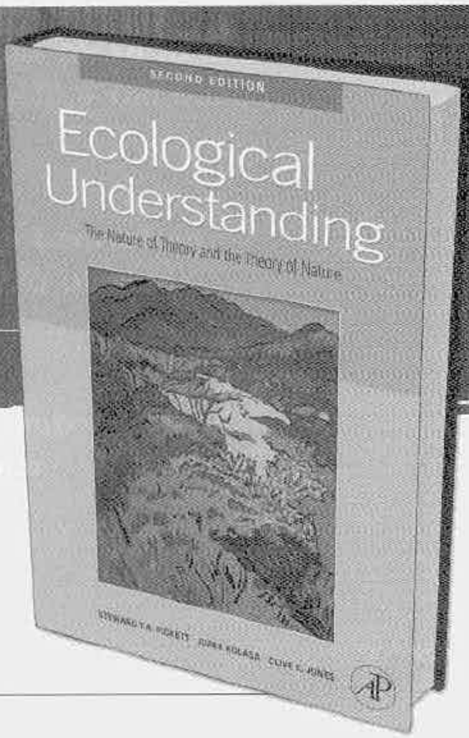
Ward, R.C., 1996. Water quality monitoring: where's the beef? Water Resources Bulletin 32, 673–680.

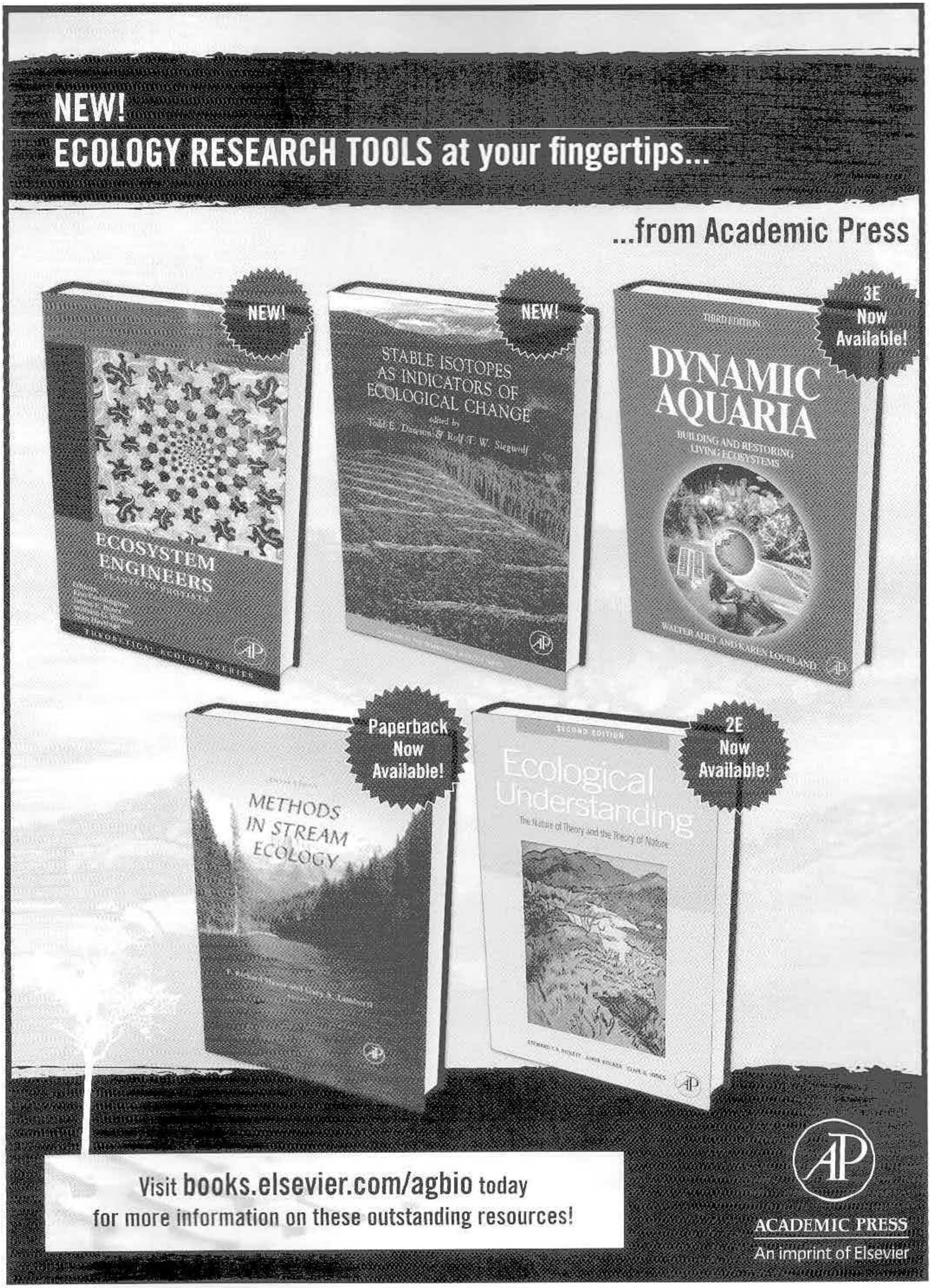
Water Framework Directive, 2003. Water Framework Directive, Common Implementation Strategy Working Group 2.7. Monitoring: guidance on monitoring for the water framework directive (2003). Final version 23 January 2003.

Woolfenden, L.R., 1984. Geohydrology and ground-water quality monitoring network for the Lower Mojave River valley, California. USGS Water-Resources Investigations Report 83-4148, USGS, Denver, CO.

World Meteorological Organization, 1994. Guide to Hydrological Practice. WMO-No. 168, WMO, Geneva.




SECOND EDITION
Ecological Understanding
The Nature of Theory and the Theory of Nature
STEWARD T.A. PICKETT
JUREK KOLASA
CLIVE G. JONES



NEW!
ECOSYSTEM ENGINEERS
PLANTS TO PROTISTS
THEORETICAL ECOLOGY SERIES
NEW!
STABLE ISOTOPES AS INDICATORS OF ECOLOGICAL CHANGE
edited by Todd E. Dawson & Rolf T. W. Siegwolf
3E Now Available!
THIRD EDITION
DYNAMIC AQUARIA
BUILDING AND RESTORING LIVING ECOSYSTEMS
WALTER ADEY AND KAREN LOVELAND
Paperback Now Available!
METHODS IN STREAM ECOLOGY
2E Now Available!
SECOND EDITION
Ecological Understanding
The Nature of Theory and the Theory of Nature
AP

Journal of

Environmental Management

June 2008 Volume 87 Number 4

Contents

Microbial and Nutrient Contaminants of Fresh and Coastal Waters

Guest Editor: Peter Hooda

Abstracted/indexed in Gale Group Business A.R.T.S., Periodical Abs Plustext, Zoological Record Online®, Science Citation Index, Science Citation Index Expanded, CC/Agriculture, Biology & Environmental Sciences, BIOSIS Previews, Biological Abstracts, GEOBASE®, Pascal, Embase, Environmental Sciences, CAB Abstracts, Social Sciences Abstracts, Elsevier BIOBASE, CSA Life Sciences Abstracts, Uncover, Medline®, Pollution Abstracts, Agricola, Biosis Toxline®, CSA Technology Research Database, ToxFile. Also abstracted/indexed in the abstract and citation database SCOPUS®. Full text available on ScienceDirect®.

Note to contributors. Online submission is available for this journal; to submit your article online, go to http://ees.elsevier.com/jema. Instructions on how to prepare a manuscript to the journal can be found in the Guide for Authors on the journal homepage at http://www.elsevier.com/locate/jenvman.

This journal is printed on acid free paper

0301-4797(200806)87:4;1-H