Grassland Productivity and Water Quality: A 21st Century Issue

David M. Nash
Department of Primary Industries, Australia

P. M. Haygarth
Institute of Grassland and Environmental Research, UK

The XX International Grassland Congress took place in Ireland and the UK in June-July 2005. The main congress took place in Dublin from 26 June to 1 July and was followed by post congress satellite workshops in Aberystwyth, Belfast, Cork, Glasgow and Oxford. The meeting was hosted by the Irish Grassland Association and the British Grassland Society.

Proceedings Editor: D. A. McGilloway

Publisher: Wageningen Academic Publishers, The Netherlands

Grassland productivity and water quality: a 21st Century issue

D.M. Nash[1] and P.M. Haygarth[2]
[1]Primary Industries Research Victoria (PIRVic), Department of Primary Industries-Ellinbank Centre, RMB2460 Hazeldean Road, Ellinbank Victoria, Australia 3821
Email: david.nash@dpi.vic.gov.au
[2]Institute of Grassland and Environmental Research, Soil Environmental and Ecological Sciences, North Wyke Research Station, Okehampton, Devon, EX20 2SB, UK

Key points

1. Irrigation and other changes to the hydrological cycle can increase soil and water salinity.
2. Primary salinisation is a natural process that affects much of Europe, Asia, Africa, the Americas and Australia. Secondary salinisation is caused by human activities such as irrigation and land clearing that mobilise salt stored in the soil.
3. The critical water contaminants exported from grasslands are nitrogen, phosphorus, potential pathogens and sediment.
4. The mechanisms responsible for diffuse pollution from grasslands and mitigation strategies are most effectively investigated using a 'source-mobilisation-transport' framework.
5. There is a lack of coherent interaction across discipline boundaries that links pollutant sources to impact. Grassland scientists need to work hand-in-hand with hydrologists and limnologists, to understand the water flows and the intricacies of ecological response, in stream or lake, in order to achieve a more coordinated and inclusive, holistic platform of research.

Keywords: salinisation, nitrogen, phosphorus, sediment, pathogens

Introduction

Grass and forage production systems are major users and exporters of water. For example, of ca. 3000 km^3 of freshwater used by agriculture, over 80% is applied to more than 250 million hectares of irrigated crops and pastures. Grasslands modify global hydrological cycles and influence the quality of the water that passes through them. This paper will consider 1) the effects of water quality on grassland and forage production, with particular emphasis on salinisation and 2) the wider effects of grassland farming on water quality for aquatic ecosystems and for other uses.

The effect of water quality on grassland and forage production, with particular emphasis on salinisation

To meet plant water needs for food and fibre production, in many parts of the world, irrigation is required to supplement natural rainfall. While primary salinisation is a natural process affecting ca. 955 Mha of Europe, Asia, Africa, the Americas and Australia, irrigation and man-made changes to the hydrological cycle have the potential to increase soil and water salinity in a process commonly referred to as secondary salinisation (Ghassemi *et al*., 1995). Often, the consequential process of sodification (alkalinsation) in which clays become increasingly saturated with sodium ions exacerbates secondary salinisation, adversely affecting soil physical properties such as structure and infiltration rates. It follows that secondary salinisation is often associated with water logging (Ghassemi *et al*., 1995).

Saline soils adversely affect plant growth by establishing an osmotic gradient that favours desiccation of plant roots (i.e. osmotic effect) and through specific ion toxicities. Some plant species have adapted to growing in saline conditions by excluding salt at the root surface, removing salt from their cytoplasm and placing it in vacuoles (where few metabolic processes occur) and excreting salt from their leaves. All three mechanisms are prevalent in highly salt tolerant plants (halophytes) (Barrett-Lennard, 2003), which can help ameliorate the effects of salinity (Glenn *et al*., 1999). In Australia, for example, revegetation of saline land with salt tolerant forage species has increased the carrying capacity of grassland, lowered water tables and lowered soil salinity (Barrett-Lennard, 2003). There is reasonable literature on grassland and forage production on saline soils (Abrol *et al*., 1988; Anon., 2004).

The quality of water supplies

While the proportion of freshwater used by agriculture worldwide is <3% of that available for ecosystem and human services, gaining access to good quality water for forage production is becoming increasingly difficult. The result has been a shift towards the use of lower quality waters, particularly saline drainage and groundwater, and industrial and municipal wastewaters (Al-Attar, 2002).

Grassland and forage production using saline irrigation water

Irrigation using saline water has been extensively studied (Richards, 1954). Limitations on the use of saline water for irrigation include the accumulation of salts in the root zone, excessive concentrations of phyto-toxic elements such as sodium, chloride, selenium, molybdenum and boron in the soil, and increased percentages of sodium adsorbed by clays.

Evapotranspiration of saline irrigation water from soils increases salt concentrations in the root zone of crops, adversely affecting their productivity. It follows that the 'salinity hazard' or risk to productivity increases with the salt concentration of irrigation water and irrigation using saline water requires that additional water beyond the needs of the crop (a leaching fraction) be applied in order to remove accumulated salts. Considerable effort has been devoted to calculating the leaching fraction (i.e. proportion of irrigation water draining below the root zone) required to maintain satisfactory growing conditions (Richards, 1954). Where drainage is restricted (i.e. clay soils) the hazard of using water of particular salinities generally increases. Consequently, irrigation water classification systems often consider both the salt concentration of the water and the characteristics of the soil to which it is applied (Table 1).

Table 1 General salinity guidelines for irrigation water[1]

Water	EC[2] (dS/m)	TDS[3] (mg/l)	Application
Fresh	<0.7	<450	No restrictions on use, potable, all crops
Brackish	0.7-3.0	450-2000	Slight to moderate restrictions on use, livestock water, most crops
Saline	>3.0	>2000	Severe restrictions on use, salt-tolerant crops, not used on soils with restricted drainage.

[1]adapted from Ayers & Westcot, 1994; [2]Electrical conductivity; [3]Total dissolved salts

Plants vary in their tolerance of salt, and as a consequence, of the salinity in irrigation water. Grasses and fodder crops tend to be amongst the more tolerant plants, although there are species and cultivar differences. For example, *Meticago sativa* (lucerne), *Agrostis stolonifera palustris* (bent grass), most *Trifolium* spp. (clovers) and *Zea Mays* (corn) are moderately sensitive to salt (i.e. 20% yield decline at >2.5 EC_e dS/m), whereas *Phalaris arundinacea* (canary grass), *Festuca* spp. (fescues), *Brassica napus* (rape) and most *Triticum* spp. (wheats) are moderately tolerant to tolerant (i.e. 20% yield decline at >10 or 14 EC_e dS/m respectively) (Maas & Hoffman, 1977; Maas, 1996).

Irrigation waters often contain nutrients such as chloride, sodium and boron that are essential for plant growth, but at higher concentrations are potentially toxic. As irrigation water constituents vary depending on the water source, the risks associated with specific ion toxicities need to be assessed on a case-by-case basis (Ayers & Westcot, 1994; Maas, 1996).

Salt in the root zone also affects soil structure. Compared to calcium and magnesium, sodium adsorption increases the tendency of clay aggregates to swell and disperse. The Exchangeable Sodium Percentage (ESP) is related to the Sodium Adsorption Ratio (SAR, Equation 1) of the soil solution (or applied water). At the ESP range most common in agricultural soils (ESP<30), the numerical values of ESP and SAR are almost equal. The 'sodium hazard' of irrigation water is therefore related to the concentration of sodium relative to calcium and magnesium concentrations, expressed by the SAR. It is of note that where salts are concentrated by evapotranspiration and as a result, the proportions of sodium, calcium and magnesium in solution are preserved, the SAR and sodium hazard of that water increases. Consequently, some authors have proposed leaching requirements to assist with sodium management (Rhoades, 1968).

$$\mathrm{SAR} = [Na^+]/\sqrt{(0.5([Ca^{2+}]+[Mg^{2+}]))} \qquad (1)$$

Where $[Na^+]$ = Sodium concentration (activity) in the solution in meq/l
$[Ca^{2+}]$ = Calcium concentration (activity) in the solution in meq/l
$[Mg^{2+}]$ = Magnesium concentration (activity) in the solution in meq/l

Even small changes in the ESP (i.e. <5%) can adversely affect soil physical properties. As the SAR of irrigation water increases, sodium is initially adsorbed to the surface of clay domains (i.e. quasi-crystals, groups of clay platelets) within aggregates. This increases the tendency for water to penetrate the spaces between the domains, increasing the distance between the domains and adversely affecting the ability of short-range attractive forces to hold the domains together, resulting in potentially dispersive behaviour. With further increases in ESP, sodium can penetrate the domains themselves increasing electrostatic repulsion between the platelets. As a result the clays swell and disperse. This phenomenon of 'demixing' is thought to account for dispersion being the dominant process at low ESP values (<15-25) and swelling at high ESP values.

The effects of a particular SAR/ESP depend on soil properties including textural class (i.e. clay percentage), clay type, quantity and type of entities that stabilise clay aggregates (i.e. iron and aluminium or organic matter). By far the most important factor affecting the expression of a particular ESP is the electrolyte concentration in the soil water. Electrostatic repulsion between moieties is greatly increased by lower electrolyte concentrations. Consequently, it is only when high quality irrigation water (i.e. low salt) is applied or rainfall occurs that the full effects of adsorbed sodium are expressed in the form of swelling and dispersion.

The effects of adsorbed sodium are particularly important in grassland and forage production systems where physical stresses placed on the soil by animal and vehicular traffic, cultivation and the like can exacerbate adverse changes in soil structure, especially where soils are waterlogged. Increased organic matter can often lessen such effects (Nelson *et al.*, 1999).

Wastewater irrigation

Industrial and municipal effluents and agricultural drainage (wastewaters) are important sources of water and nutrients. With increasing urbanisation and the associated demand for potable water, municipal (i.e. primarily domestic) wastewaters, in particular, are a useful substitute for freshwater that would otherwise be used for irrigation. Due to the highly variable quality arising from source differences and the treatment prior to land application, wastewater use in agriculture has not been without peril for either agricultural workers or consumers of the agricultural produce. Guidelines have been developed to protect against such occurrences (Blumenthal *et al.*, 2000). In addition, wastewater irrigation carries the risk of introducing potentially toxic substances, including heavy metals, into the human food chain and microbiological health risks (Mara & Caincross, 1989).

Many studies have reported decreases in soil infiltration rates and hydraulic conductivities, following wastewater irrigation. Some of the mechanisms that could be responsible for these changes include solvent-solute effects on clays (Anandarajah, 2003), accumulation of suspended solids at the soil surface or blockage of the inter-soil spaces by suspended material such as colloidal clay and cells from microorganisms (Bouwer & Chaney, 1974; Metzger *et al.*, 1983), entrapped air (Rice, 1974), the formation of a biological mat or crust including the production of microbial extracellular polymeric materials such as polysaccharides (Balks *et al.*, 1997; Taylor & Jaffe, 1990), collapse of soil structure due to organic matter dissolution (Lieffering & McLay, 1996) and physico-chemical changes to pore geometry and micro-fabric that are related to sodicity, cation exchange reactions and exchange hysteresis (Shainberg & Letey, 1984).

The swelling and dispersion of clay aggregates is likely to be particularly important in decreasing soil infiltration rates as many wastewater constituents increase the effective SAR of irrigation water. This can arise in various ways. During wastewater irrigation inorganic anions, especially SO_4^{2-}, CO_3^{2-}, HCO_3^{-} precipitate Ca^{2+} and Mg^{2+} from solution (Suarez, 1981), organic matter can lower the activity of Ca^{2+} and Mg^{2+} and cations that would otherwise form bridges between minerals, due to changes in soil pH, for example, the capacity of organic matter to bind particles can decrease and anaerobic or other conditions can be created that result in the chemical reduction of species that would otherwise form bridges between minerals (i.e. Fe^{3+}) (Reid *et al.*, 1982; Shainberg & Letey, 1984; Visser & Caillier, 1988; Piccolo & Mbagwu, 1989). The ability of wastewater constituents to act synergistically and impair the longer-term productive capacity of grassland production systems warrants further investigation.

Contaminants are an additional risk associated with the use of wastewater for irrigation. Some contaminants, such as zinc and copper, can accumulate to phytotoxic concentrations (Abdelrahman & Al-Ajmi, 1994). However, of greater concern are wastewater contaminants that while not phytotoxic, may be introduced into the human food chain via grassland or fodder through the grazing animals (Cooper, 1991). These include heavy metals such as cadmium and various organic chemicals that can be classed as endocrine disruptors (i.e. DDT, dieldrin, endosulfin) and biological toxins (i.e. algal toxins). These contaminants can access the human food chain in forage or as a contaminant of forage, or by the ingestion of contaminated soil by grazing animals. There is a pressing need to understand the risks associated with wastewater contaminants and develop preventative measures that allay consumer concerns.

The effects of grassland farming on water quality

Water exported from grassland production systems often contains pollutants such as sediment nitrogen, phosphorus, labile organic materials and microorganisms. For a pollutant to pose a threat to water quality there needs to be (1) a source of the pollutant that is (2) mobilised into water and (3) transported to a location where it has (4) an adverse impact. The first three of these components are often associated with grassland production systems and a conceptual framework for characterising pollutant exports is presented in Figure 1. Most pollutant exports are episodic and underlying pollutant sources and mobilisation and transport processes are 'incidental' factors associated with climate and management. This framework is useful in that it breaks down diffuse pollution into a simple *source*, *mobilisation* and *transport* framework, highlighting the importance of all stages in the diffuse pollution 'continuum' (Haygarth *et al*., 2005). This approach can help us focus on the real cause of the problems, where intervention is most likely to be effective, and help prioritise potential mitigation strategies.

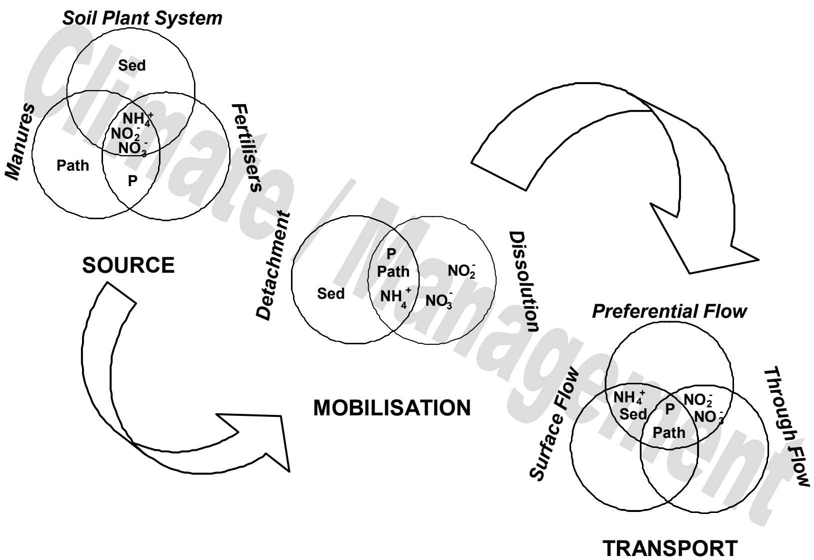

Figure 1 A conceptual framework for characterising diffuse pollutant exports from grasslands (Sed = sediment, Path = potential pathogens, NH_4^+ = ammonium, NO_3^- = nitrate, NO_2^- = nitrite, P = phosphorus) (adapted from Haygarth *et al*., 2004)

In grassland systems, pollutant sources include agrochemicals such as fertilisers, herbicides and pesticides, the soil fabric and organic matter, live and decomposing vegetation, animals and animal defecation products deposited direct to pasture or in various forms of manure and slurry that are collected and subsequently redistributed on the land. Critical incidents such as fertiliser or manure application can increase pollutant availability and, as a consequence, exports. These 'incidental' exports are often 'preventable' and source modification has traditionally been the focus of much research effort. However, unless all pollutant sources are eliminated, even with optimal management pollutants will be exported from grassland production systems. This 'systematic' component of pollutant exports has received little study compared with the 'incidental', and often 'preventable' component. In part this is because factors such as manure application rates and timing are readily examined with standard experimental techniques, as the treatment effects are often large compared with the variability between replicates.

Pollutants can be mobilised either as particulate (>0.45µm) or dissolved (<0.45µm) materials. Particulate materials and the pollutants they may contain are mobilised by predominantly physical processes that detach sediment from the soil fabric. Dissolution (sometimes otherwise referred to as 'solubilisation' (Haygarth *et al*., 2005)) of pollutants, on the other hand, is largely a biochemical, and often time dependant process. Again critical incidents ('incidentals') can enhance pollutant mobilisation. These effects are most pronounced for detachment processes where physical perturbation of the land (i.e. poaching and pugging of the land surface and overgrazing) can enhance the mobilisation of particulate pollutants. River siltation is a direct consequence of some grassland production systems. Dissolution processes being subtle, are generally considered less effected by 'incidentals'. However, management that changes soil properties, such as pH, may well affect dissolution processes.

Pollutants can be transported in both surface and subsurface pathways. The effectiveness of the transport process will depend on the pollutant in question and the particular pathway (Nash *et al*., 2002). Transport processes are particularly susceptible to incidental factors, especially climatic variables such as rainfall intensity and management decisions such as the installation of drainage that increases the hydrological connectivity between the source and receiving waters. Many strategies have been used to interrupt the transport of pollutants originating from grassland production systems. These include agricultural drainage recycling systems and grassed buffer strips and other installations that lower water velocity (and kinetic energy), facilitating the sedimentation of particulate materials.

In terms of preventing unacceptable water pollution, understanding the interactions between pollutant sources, mobilisation and transport processes and how critical incidents ('incidentals') affect these relationships offers our best opportunity for developing effective remedial strategies. A good example is detached sediment. Detached sediments can 'adsorb' dissolved pollutants in transit (Sharpley *et al*., 1981) and through their deposition or entrapment, remove dissolved pollutants from the water column. A number of studies has shown that methods used to decrease detachment and transport of sediment actually increased the concentrations of dissolved, and in some cases more potent forms of the same pollutant in the water. There is therefore a clear need for a multi-dimensional approach to this research.

Phosphorus

Elevated concentrations of phosphorus in streams, rivers and lakes may contribute to eutrophication (Pierzynski *et al*., 2000). Eutrophication limits water use for drinking, fishing, industry and recreation (Carpenter *et al*., 1998). Dissolved phosphorus concentrations have increased markedly in recent decades in Australia, Europe, New Zealand, and the USA. Such changes have been linked to large-scale fish kills in estuaries caused by increased populations of the dinoflagellate *Pfiesteria piscicida*, which in turn have the potential to adversely affect human health (Burkholder *et al*., 1992). The European Union Water Framework Directive (2000/60/EC) aims to restore all waters to 'good ecological status' by 2015.

Many of the current remedial strategies rely on decreasing source availability, especially through improved management of phosphorus in manures, slurries and fertilisers.

Mobilisation control measures have generally focussed on stopping detachment processes by increasing groundcover, improving soil physical properties and minimising incidentals by ensuring management decisions take account of the prevailing climatic conditions. Transport (sometimes called 'delivery') control measures focus on preventing the movement of mobilised phosphorus across the landscape. They include the use of buffers, wetlands and drainage recycling systems. Although delivery control measures can produce a reasonable decrease in phosphorus loads, these measures only have a limited lifetime and generally require 'landscape wide' application that is often difficult to achieve in practice. Overviews of phosphorus and water quality issues relevant to grasslands can be found in Leinweber *et al*. (2002) and Haygarth *et al*. (2005).

Nitrogen

The various forms of nitrogen affect water quality differently. Nitrate is the most prevalent soluble form affecting freshwater and has generally been considered the most 'troublesome' contributing to acidification of waters and eutrophication, both of which result in major changes to aquatic bio-diversity and community structure. In potable water, nitrate may be associated with methaemoglobinaemia in very young infants and with stomach cancer, although more recently, there is some controversy over the extent of these associations (Addiscott, 1999).

Though monitored less often than nitrate, nitrite also adversely affects receiving waters. Nitrite is highly toxic to fish and invertebrates because it impairs their ability to take up oxygen and bonds with haemoglobin, thereby decreasing the oxygen carrying capacity of the blood. The European Union Freshwater Fish Directive guidelines for salmonid and coarse fish are 3.0 and 9.0 μg nitrite-N/l, respectively (European Community, 1978), which are below the concentrations in many grassland farming catchments (Haygarth *et al*., 2004).

Ammonium transfers to watercourses can also affect water quality. While dissolved ammonia is directly toxic to fresh water fish, the transformation (nitrification) of ammonium to nitrate in water can contribute to oxygen depletion (Haygarth *et al*., 2004).

Mitigation of nitrogen exports from grasslands has focused on source and transport control, especially in winter. Source control options are centred on maintaining the efficiency of plant uptake and trying to avoid excess application. The latter is best achieved through application of fertiliser in regular but small amounts, strategic use of fertiliser in relation to available mineral nitrogen (this includes soil testing), seeking to breed for plants that are more efficient and have deeper rooting depth, and improving efficiency and thus nitrogen requirements of the grazing animal (Scholefield *et al*., 1993). Transport based mitigation options are to avoid drainage of land because this increases both aeration of the soil and mineralization of nitrogen, and enhances hydrological transport. Sandy textured soils need to be particularly carefully managed. Further information on nitrogen cycling and potential mitigation is provided in Scholefield *et al*. (1993) and Hatch *et al*. (2002).

Sediment

Sediment has for a long time been 'passed over' as a minor water quality problem associated with agricultural soils and grasslands in particular. However, the transfer of sediment from grasslands is one of the most significant water quality and stream health issues. For example, siltation of gravel-bed rivers prevents spawning of salmonids to the extent that reproduction is severely impaired and salmon populations in key areas of the world (e.g. South West England and Mississippi USA) have declined, with consequential impacts on ecological balance and recreational fishing.

Sediment transfers from agricultural land arise from poaching (grazing stock treading on the surface of moist or wet soil) (McDowell *et al.*, 2003) and traffic by both animals and machinery (Harrod & Theurer, 2002). Such perturbations reduce infiltration, increasing the potential for overland flow of water. The outcome is that water flow concentrates on the surface and at times of intense runoff, may become channelised with the resulting increase in kinetic energy detaching soil particles. Additional problems arise because sediment carries contaminants, particularly phosphorus and pathogens. It follows that there are mechanistic overlaps between the mitigation measures used to prevent the mobilisation of sediment and particulate (detached) phosphorus. Such mitigation measures try to avoid scenarios that give rise to compaction of the grassland surface, especially over wintering and a general presence of animals or traffic at times when the surface is saturated and vulnerable to reduced infiltration. More information on sediment transfers and impacts from grasslands is presented in Harrod & Theurer (2002).

Potential pathogens

The rumen and digestive tract of agricultural livestock is host to a diversity of microflora and can act as a reservoir for micro-organisms (Rasmussen *et al.*, 1993) that may be pathogenic to humans. As a result organisms such as *E. coli* 0157, *Salmonella* spp., *Campylobacter jejuni*, *Listeria monocytogenes*, *Cryptosporidium parvum* and *Giardia intestinalis* may contaminate livestock wastes (Oliver *et al.*, 2005). Contamination of grassland surfaces can result from either direct defecation from livestock or as a result of spreading recycled manures. Since many of these organisms persist in manure and in soil, they are vulnerable to transfer to water where they can present a serious threat to humans.

Opportunities for decreasing pathogen transfer to water include controlling pathogen sources and lessening the persistence (increasing the 'die off') of the organisms in soil and manure, in combination with minimising transport factors. Pathogen transfer is one of the least studied and emerging issues for grasslands and water quality, but presents many problems and challenges for the 21^{st} century because of the potential to *directly* affect the health of human beings. Relevant reviews on this subject can be found in Jones (2002), and Oliver *et al.* (2005).

Conclusions and recommendations for future research

As the demand for livestock and livestock production continues to rise, grassland productivity and water quality will become truly 21^{st} Century issues. There are many water quality problems that emanate from grassland production and to a large extent their seriousness and consequences have yet to be fully acknowledged. Nitrogen, because of its sheer volume of usage, is the most researched and understood contaminant. However, its impact in some regions of the world is now thought to be of less importance than phosphorus, which controls the productivity of many inland freshwater lakes and waterways. Sediment and pathogen transfers represent the 'newer' challenge for the 21^{st} century. Many of the processes and mitigating strategies, particularly for micro-organisms, lack mechanistic information and further research into pathogen survival and mitigation is required.

In future, grassland research will also need to embrace a wider and 'scaled-up' view of water quality problems, with greater emphasis on landscape 'delivery' and transport controls (Beven *et al*., 2004), to balance the relatively large research effort on finer scale processes that has been the priority for many years. There is a pressing need to integrate results from these finer scale process studies into mathematical and conceptual frameworks on which policy decisions can be based. Software that combines probability distributions through various simulation procedures (i.e. Monte Carlo simulations) and Bayesian networks are tools that can be used, in conjunction with, for example, source decay (i.e. the availability of pollutants from fertiliser with time after application) to investigate the probability that a particular mitigation strategy will achieve its objectives.

There are also other pressing deficiencies in research that must be addressed in order for progress to be made. The first is the lack of coherent interaction across discipline boundaries that links source to impact. Grassland scientists need to work hand-in-hand with hydrologists and limnologists, to understand the water flows and the intricacies of ecological response, in stream or lake, in order to achieve a more coordinated and inclusive, holistic platform of research. Secondly, although not to be treated separate to the previous issue, is the urgency for the testing of mitigation options and best management practices (BMPs) in more coordinated and localised platforms. It is not known quantitatively, how effective such management practices will be, though their performance is sure to differ in different catchments around the world. It will also be necessary to consider the economic viability of such options and offset mitigation effectiveness with potential cost effectiveness. There remains much to do.

While grasslands contribute to water quality problems they can also be part of the solution. There is considerable scope for developing irrigation systems with a reduced environmental footprint capable of mitigating past mistakes. For example, 'conjunctive water use', the mixing of saline ground water with higher quality (i.e. lower salt) waters has been used to lower water tables in irrigation regions (Prendergast *et al*., 1994; Jury *et al*., 2003) and, to a lesser extent, alleviate the effects of adsorbed sodium. 'Serial biological concentration' (Heuperman, 1999; Jury *et al*., 2003) has been used to concentrate pollutants, including nutrient and other salts, which would otherwise have been released into the aquatic environment. Adapting and optimising systems for different climatic and physical environments, and social and cultural systems will be important if the quantity of water available for grassland and fodder production is to increase, the environmental impact of altered hydrology and the associated secondary salinisation is minimised, and off-site water quality is improved.

Acknowledgements

The Institute of Grassland and Environmental Research (IGER) is in part supported by the UK Biotechnology and Biological Sciences Research Council (BBSRC). Defra projects ES0121, NT2511 and PE0118 and the associated teams. Others to whom thanks are due include Philomena Gangaiya, Fiona Robertson, Kirsten Barlow and Will Gates (DPI) and Louise Heathwaite (University of Lancaster). The contributions of the Department of Primary Industries to this paper were supported by the Victorian Government, Dairy Australia and GippsDairy.

References

Abdelrahman, H.A., & H. Al-Ajmi (1994). Heavy metals in some water- and wastewater- irrigated soils of Oman. *Communications in Soil Science and Plant Analysis*, 25, 605-613.

Abrol, I.P., J.S.P. Yadav & F.I. Massoud (1988). Salt-affected soils and their management. Food and Agriculture Organisation of the United Nations, Rome, Italy, 131pp.

Addiscott, T.M. (1999). Nitrate and health: introductory comments. In: R.H. Hinton (ed.) Managing risks of nitrates to humans and the environment. The Royal Society of Chemistry, Cambridge, 247-299.

Al-Attar, M. (2002). Role of biosaline agriculture in managing freshwater shortages and improving water security. Paper presented at the World Food Prize Symposium 'From Middle East to Middle West: Managing Freshwater Shortages and Regional Water Security',Des Moines, USA, October 24-25.

Anandarajah, A. (2003). Mechanism controlling permeability change in clays due to changes in pore fluid. *Journal of Geotechnical and Geoenvironmental Engineering*, 129, 163-172.

Anonymous (2004). Biosalinity awareness project. Understanding the impact of salinisation and implications for future agriculture. Biosalinity Awareness Project <www.biosalinity.org>

Ayers, R.S. & D.W. Westcot (1994). Water quality for agriculture. Food and Agriculture Organization of the United Nations, Rome, Italy, 173pp.

Balks, M.R., C.D.A. McLay C.G. & Harfoot (1997). Determination of the progression in soil microbial response, and changes in soil permeability, following application of meat processing effluent to soil. *Applied Soil Ecology*, 6, 109-116.

Barrett-Lennard, E.G. (2003). Saltland pastures in Australia - a practical guide. Land, water and wool sustainable grazing on saline land project. Land and Water Australia, Canberra, Australia, 176pp.

Beven, K.B., L. Heathwaite, P.M. Haygarth, D. Walling, R. Brazier & P. Withers (2004). On the concept of delivery of sediment and nutrients to stream channels. *Hydrological Processes*, (in press).

Blumenthal, U.J., A. Peasey, G. Ruiz-Palacios & D.D. Mara (2000). Guidelines for wastewater reuse in agriculture and aquaculture: recommended revisions based on new research evidence. Loughborough University, Loughborough, 87pp.

Bouwer, H. & R.L. Chaney (1974). Land treatment of wastewater. *Advances in Agronomy*, 26, 133-176.

Burkholder, J.M., E.J. Noga, C.W. Hobbs, H.B. Glasgow Jr. & S.A. Smith (1992). New 'phantom' dinoflagellate is the causative agent of major estuarine fish kills. *Nature*, 358, 407-410.

Carpenter, S.R., N.F. Caraco, D.L. Correll, R.W. Howarth, A.N. Sharpley & V.H. Smith (1998). Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecological Applications*, 8, 559-568.

Cooper, R.C. (1991). Public health concerns in wastewater reuse. *Water Science and Technology*, 24, 55-65.

European Community (1978). Council directive of 18 July 1978 on the quality of fresh waters needing protection or improvement in order to support fish life. *78/659/EEC Official Journal of the European Communities*, 21, 1-10.

Ghassemi, F., A.J. Jakeman & H.A. Nix (1995). Salinisation of land and water resources: human causes, extent, management and case studies. University of New South Wales Press, Sydney, Australia, 526pp.

Glenn, E.P., J.J. Brown & E. Blumwald (1999). Salt tolerance and crop potential of halophytes. *Critical Reviews in Plant Sciences*, 18, 227-255.

Harrod, T.R. & F.D. Theurer (2002). Sediment. In: S.C. Jarvis (ed.) Agriculture, hydrology and water quality. CAB International, Wallingford, UK, 155-170.

Hatch, D., K. Goulding & D. Murphy (2002). Nitrogen. In: P.M. Haygarth & S.C. Jarvis (eds.) Agriculture, hydrology and water quality. CAB International, Wallingford, UK, 7-18.

Haygarth, P.M., D.C. Chadwick, S. Granger, B. Chambers, S. Anthony, K. Smith & D. Harris (2004). Characteristic behaviour and potential mitigation of some diffuse pollutants in England and Wales: a review of biological oxygen demand, ammonium, nitrite and some potential pathogens. Institute of Grassland and Environmental Research, Okehampton, Devon, 124pp.

Haygarth, P.M., L.M. Condron, A.L. Heathwaite, B.L. Turner & G.P. Harris (2005). The phosphorus transfer continuum: linking source to impact with an interdisciplinary and multi-scaled approach. *Science of the Total Environment*, (in press).

Heuperman, A. (1999). Serial biological concentration: a new management system for saline drainage water. *Irrigation Australia*, 14, 21-22, 27.

Jones, D. (2002). Human enteric pathogens. In: P.M. Haygarth & S.C. Jarvis (eds.) Agriculture, hydrology and water quality. CAB International, Wallingford, UK, 133-153.

Jury, W.A., A. Tuli &J. Letey (2003). Effect of travel time on management of a sequential reuse drainage operation. *Soil Science Society of America Journal*, 67, 1122-1126.

Leinweber, P., B.L. Turner & R. Meissner (2002). Phosphorus. In: P.M. Haygarth & S.C. Jarvis (eds.) Agriculture, hydrology and water quality. CAB International, Wallingford, UK, 29-56.

Lieffering, R.E. & C.D.A. McLay (1996). The effects of strong hydroxide solutions on the stability of aggregates and hydraulic conductivity of soil. *European Journal of Soil Science*, 47, 43-50.
Maas, E.V. (1996). Crop salt tolerance. In: K.K. Tanji (ed.) Agricultural salinity assessment and management. American Society of Civil Engineers. New York, 262-304.
Maas, E.V. & G.J. Hoffman (1977). Crop salt tolerance - current assessment. Journal of the Irrigation and Drainage Division, *Proceedings of the American Society of Civil Engineers*, 103, 115-134.
Mara, D. & S. Caincross (1989). Guidelines for the safe use of wastewater and excreta in agriculture and aquaculture: measures for public health protection. World Health Organization and UNEP, Geneva, 74pp.
McDowell, R.W., J.J. Drewery, R.W. Muirhead & R.J. Paton (2003). Cattle treading and phosphorus and sediment loss in overland flow from grazed cropland. *Australian Journal of Soil Research*, 41, 1521-1532.
Metzger, L., B. Yaron & U. Mingelgrin (1983). Soil hydraulic conductivity as affected by physical and chemical properties of effluents. *Agronomie*, 8, 771-778.
Nash, D.M., D. Halliwell & J. Cox (2002). Hydrological mobilisation of pollutants at the slope/field scale. In: P.M. Haygarth & S.C. Jarvis (eds.) Agriculture, hydrology and water quality. CAB International, Wallingford, UK, 225-242.
Nelson, P.N., J.A. Baldock, P. Clarke, J.M. Oades & G.J. Churchman (1999). Dispersed clay and organic matter in soil: their nature and associations. *Australian Journal of Soil Research*, 37, 289-315.
Oliver, D.M., C.D. Clegg, P.M. Haygarth &A.L. Heathwaite (2005). Assessing the potential for pathogen transfer from grassland soils to surface waters. *Advances in Agronomy*, 85, 125-180.
Piccolo, A., &J.S.C. Mbagwu (1989). Effect of humic substances and surfactants on the stability of soil aggregates. *Soil Science*, 147, 47-54.
Pierzynski, G.M., J.T. Sims & G.F. Vance (2000). Soils and environmental quality. CRC press, Boca Raton, London, New York, Washington DC, 459p.
Prendergast, J.B., C.W. Rose & W.L. Hogarth (1994). A model for conjunctive use of ground water and surface waters for control of irrigation salinity. *Irrigation Science*, 14, 167-175.
Rasmussen, M.A., W.C. Cray, T.A. Casey & S.C. Whipp (1993). Rumen contents as a reservoir of enterohemorrhagic *Escherichia coli*. *FEMS Microbiology Letters*, 114, 79-84.
Reid, J.B., M.J. Goss & P.D. Robertson (1982). Relationship between the decreases in soil stability effected by the growth of maize roots and changes in organically bound iron and aluminium. *Journal of Soil Science*, 33, 397-410.
Rhoades, J.D. (1968). Leaching requirement for exchangeable sodium control. *Soil Science Society of America Journal*, 32, 652-656.
Rice, R.C. (1974). Soil clogging during infiltration of secondary effluent. *Journal of Water Pollution Control Federation*, 46, 708-716.
Richards, L.A. (1954). Diagnosis and improvements of saline and alkaline soils. Agriculture Handbook 60. USDA, Washington, DC, 160p.
Scholefield, D., K.T. Tyson, E.A. Garwood, A.C. Armstrong, J. Hawkins & A.C. Stone (1993). Nitrate leaching from grazed grassland lysimeters: effects of fertilizer input, field drainage and patterns of weather. *Journal of Soil Science*, 44, 601-614.
Shainberg, I., & J. Letey (1984). Response of soils to sodic and saline conditions. *Hilgardia*, 52, 1-57.
Sharpley, A.N., R. Menzel, S. Smith, E. Rhoades & A. Olness (1981). The sorption of soluble phosphorus by soil material during transport in runoff from cropped and grassed watersheds. *Journal of Environmental Quality*, 10, 211-215.
Suarez, D.L. (1981). Relation between pH_c and sodium adsorption ratio (SAR) and an alternative method of estimating SAR of soil or drainage water. *Soil Science Society of America Journal*, 45, 469-475.
Taylor, S.W. & P.R. Jaffe (1990). Biofilm growth and the related changes in the physical properties of a porous medium. 1. Experimental investigation. *Water Resources Research*, 26, 2153-2169.
Visser, S.A. & M. Caillier (1988). Observations on the dispersion and aggregation of clays by humic substances. I. Dispersive effects of humic acids. *Geoderma*, 42, 331-337.