LANDSCAPE LOGIC

LINKING LAND AND WATER MANAGEMENT TO RESOURCE CONDITION TARGETS

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Streamside management zones for buffering streams on farms: observations and nitrate modelling

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Cover photo: Two types of streamside management zones (SMZs) are shown, both of which included fences to exclude livestock. In the foreground the SMZ was planted with *Acacia melanoxylon* (blackwoods) and not intended for commercial wood production. In the background is an SMZ containing commercial 20-year-old *Eucalyptus nitens* that was harvested and reported in Neary *et al*. (2010).

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LANDSCAPE LOGIC is a research hub under the Commonwealth Environmental Research Facilities scheme, managed by the Department of Sustainability, Environment, Water, Population and Communities.
It is a partnership between:

- **six regional organisations** – the North Central, North East & Goulburn–Broken Catchment Management Authorities in Victoria and the North, South and Cradle Coast Natural Resource Management organisations in Tasmania;
- **five research institutions** – University of Tasmania, Australian National University, RMIT University, Charles Sturt University and CSIRO; and
- **state land management agencies in Tasmania and Victoria** – the Tasmanian Department of Primary Industries & Water, Forestry Tasmania and the Victorian Department of Sustainability & Environment.

The purpose of Landscape Logic is to work in partnership with regional natural resource managers to develop decision-making approaches that improve the effectiveness of environmental management.
Landscape Logic aims to:

1. Develop better ways to organise existing knowledge and assumptions about links between land and water management and environmental outcomes.
2. Improve our understanding of the links between land management and environmental outcomes through historical studies of private and public investment into water quality and native vegetation condition.

Streamside management zones for buffering streams on farms: observations and nitrate modelling

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Summary

Natural resource managers need quantitative information on the effectiveness of streamside management zones (SMZ) in agricultural landscapes for protecting water quality. Analysis of buffer experiments internationally had previously suggested that a buffer width of 15 m would remove about 80% of nitrogen (N). Nitrate is the main form of N of interest, but until recently there were few Australian data or model predictions available on buffer effectiveness. In 2007, a research project commenced in the Landscape Logic CERF Hub that focused on buffering a headwater stream from N contamination, with the aim of (1) quantifying the N-buffering effect at a small catchment scale, and (2) developing a model that integrated the salient processes and that potentially could be applied to other catchments. This research complimented a related project in the CRC for Forestry that included a much lower level of nitrogen monitoring.

This report summarises our progress on these two aims. Frequent measurements were made in a previously-established, steep, paired-catchment experiment with adjacent buffered and unbuffered reference headwater streams in a low-intensity grazing system. Less frequent measurements were also made in six other nearby unbuffered catchments to provide replication of the reference condition. Modelling utilised the HYDRUS model, which has wide acceptance internationally for mechanistically simulating soil water and solute processes. We also used its N module (CW2D) that was developed for simulating nitrate removal by constructed wetlands. The 10 ha buffered catchment had an area of grazed pasture (62%) low in the landscape, and the rest was native forest. Approximately 10% of the pasture was fenced around the stream to exclude stock and to allow the establishment of a forest plantation. The adjacent 4 ha catchment in the same paddock was 99% grazed pasture and cattle had free access to its stream when stock were in the paddock. Stream water from both catchments was monitored for various forms of N. No fertiliser was applied to the pasture and only a small amount of hay was used as a feed supplement. A small amount of diammonium phosphate fertilizer was buried beside each tree seedling in the plantation soon after planting. Pools and fluxes of N measured in the buffered catchment were: pasture N uptake, N mineralisation and nitrification, and N concentrations in rain water, soil water, soil leachate, and the watertable.

Between large rainfall events (storms), nitrate concentrations in stream water were low and similar to those in the watertable of the hillslope. During monitored storms, which lasted several days, nitrate-rich water in surface soil that built up during drier periods began entering the buffered stream a day or two after the storm commenced, and continued for a day or two after rain stopped, suggesting preferential flow processes. This effect, commonly referred to as a flushing effect, was most pronounced in the buffered catchment, but it was probably not related to buffering. Annually, N export was 70–90% dissolved organic N (DON), 11–18% particulate N (PN), and <5% nitrate. Total N measured in stream flow during the drought year starting May 2008 was <1 kg/ha in both catchments, but during the subsequent wet year 9 kg N/ha was measured in stream flow of the unbuffered catchment and 6 kg N/ha in the buffered catchment. Grab samples covering a 3-year period indicated that buffering substantially reduced *E. coli*, phosphate and sediment (turbidity) concentrations. Lower concentrations of ammonium and nitrate in the buffered catchment in 2009 could not be fully attributed to a buffering effect, because similar

differences in concentrations were present in 2007 before the SMZ was established.

A method was developed for pre- and post-processing of rainfall and flow data for HYDRUS that accounted for overland flow and adequately simulated early nitrate dilution during a storm (0.08 to 0.01 mg/L) followed by an increase in nitrate concentration (0.01 to 2.4 mg/L). The HYDRUS-CW2D model was used to simulate the buffered catchment for a dry year that included measured daily rainfall and resultant flows and nitrate fluxes (including denitrification). Denitrification was predicted to occur throughout the hillslope in the saturated zone, but overall there was a negligible predicted annual rate of denitrification (2.5 kg N/ha/year), which was not predicted to increase due to trees in the buffer. A hypothetical higher-denitrification, higher-stream-flow scenario was developed where higher denitrification was simulated (12.6 kg N/ha/year) as well as a decrease in nitrate-N concentration as water drained through the riparian zone. In this scenario, buffer establishment (deep roots added in the SMZ) led to no change in denitrification and increased N uptake by vegetation. In a third scenario, additional organic matter was added in the SMZ with the tree roots; under these conditions the change in N fluxes was predicted to be +4% nitrification, +4% uptake, and –71% denitrification. While many uncertainties remain about these scenarios, our modelling did not support the assertion that denitrification would increase due to the establishment of trees in the SMZ, unless trees had a negligible effect on anaerobic conditions (depth of water table) and they led to substantially increased soil organic matter concentrations, for which in-turn we found little support in the literature.

Hence, our measurements did not indicate reduced N delivery to the stream due to buffering. Modelling suggested that if an effect was to develop it would not be via increased denitrification. Most of the observed effect could be expected to be due to decreased particulate delivery that was not simulated by the model, and that an additional contribution might be expected in the longer term via increased uptake of N by vegetation. Further, the nutrient mitigation capacity of buffers might need to be rejuvenated periodically by removing nutrients contained in plant materials by careful harvesting, grazing or mowing. To enhance future modelling using HYDUS-CW2D we provide several suggestions for model development.

Contents

Acknowledgements

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Background

For the past two decades at least, researchers and practitioners have been interested in quantifying the nitrogen mitigation effects (buffering) of streamside management zones (SMZs) in the agricultural landscape. Recent reviews of international experience suggest that the effects can be substantial. For example, Mayer *et al.* (2007) summarised the nitrate removal effectiveness of 89 individual buffers in 45 published studies. These studies included various types of SMZ vegetation, width, and landscape characteristics. Effectiveness varied widely (-258% to 100% effective) and was most effective for wide SMZs (> 25 m) and surface delivery, irrespective of vegetation type. Inconsistency in the results suggested important influences of soil type, subsurface hydrology, and biogeochemistry.

Using 73 published studies, only two of which were the same as those used by Mayer *et al.* (2007) for nitrate, Zhang et al (2010) found that SMZ width explained 44% of the variation in nitrogen removal efficiency that included three forms of nitrogen (total N, ammonium and nitrate), and that treed systems were more effective than those containing grass only or grass and trees. A buffer width of 15 m was predicted to remove 80% of N entering the up-slope side of the buffer. However, all of these studies of nitrate were done at a paddock- or plot-scale, and several only included overland flow. Hence, SMZ effects on nitrate delivery to headwater streams at a catchment scale were not quantified, and at this scale subsoil, channelized flow, and in-stream processes can be important. Our study aimed to partially address this knowledge gap by measuring SMZ effects at a headwater catchment scale, by integrating overland and subsoil flow processes, and by including all forms of nitrogen.

Because the SMZ effect varies widely between situations, it would be useful to quantify the effect in a wide variety of situations and build up enough experience to reliably predict their effects at proposed sites. However, quantification of SMZ effectiveness is time consuming and expensive, especially at a catchment scale, which precludes numerous case-by-case measurements. We were therefore interested in testing or developing a mechanistic model that could be adapted to a wide variety of situations and that took account of the salient processes involved in SMZ mitigation of nitrate delivery to streams.

Instead of nitrate being delivered to a stream it can be intercepted by plant or microbial uptake and denitrification. Hence, a suitable model needed to account for the hydrology of the system and the production, transfers and transformations of nitrate. This meant that such a model would need to mechanistically simulate within-soil water and nitrogen processes and that it needed to be spatially explicit enough to represent the effects of various width SMZs and their nitrate uptake ability. Hence, a two-dimensional (hillslope) or three-dimensional (catchment) model was needed that could be used at the scale of a few hectares (i.e. headwater catchment scale).

We considered a range of potentially useful models, and were most attracted to the HYDRUS model (Šimůnek *et al*. 2008) because it could simulate highly mechanistically the within-soil behaviour of water and solutes. In a simplistic manner it could also simulate overland flow. For example, Guan *et al.* (2010) used HYDRUS to simulate water dynamics of a hillslope with preferential flow, and Hilton *et al.* (2008) used it to simulate runoff from a grass roof. In its two-dimensional application HYDRUS also includes a nitrogen module (CW2D) developed to predict nitrate removal from constructed wetlands (Langergraber and Šimůnek 2005), and it had a user-friendly interface and a high degree of spatial and temporal flexibility.

Objectives

Our objectives were to:

(1) quantify at a headwater catchment scale the buffering effects of an SMZ, particularly for nitrogen, that combined cattle exclusion and plantation establishment,

(2) adapt the HYDRUS-CW2D model to simulate the salient processes governing water and nitrogen dynamics at a hillslope scale, and thereby estimate the relative importance of denitrification and uptake for nitrate mitigation,

(3) use HYDRUS-CW2D to estimate the effect on nitrate delivery to streams of reforestation of the streamside management zone, and

(4) provide practical guidance and development recommendations for setting up and interpreting hillslope simulations using the HYDRUS-CW2D model.

Methods

Site, Treatments, and Measurements

In collaboration with the CRC for Forestry we established a paired-catchment experiment in a single paddock with a northerly aspect in a mixed grazing and native forest landscape in southern Tasmania (Photo 1, Figure 1). These catchments were part of the larger Forsters Rivulet catchment in southern Tasmania, Australia. Fertilizers were not used in the catchment for several years prior to or during the study, except during the plantation establishment phase. Grazing was conducted at a moderate stocking density (c. 2-3 head per ha) for 2-6 weeks at 2-3 month intervals. Cattle have free access to all unfenced streams. A plantation was established in 2008 in the SMZ of one of the catchments (Photo 2).

The catchments are 3.5 ha (control) and 9.8 ha (SMZ catchment) in area. The area of the SMZ was 0.6 ha which was 6% of the catchment and 10% of the pasture area in the catchment. Within the SMZ, soil outside the saturated riparian zone was cultivated by a 'scoop-and-pile' method using a mini-excavator, which created a pit adjacent to a mound on which a tree seedling was planted. Tree seedlings were planted in August 2008. In the lower, northern half of the SMZ, *Eucalyptus nitens* (shining gum) was planted on each mound, and *Acacia melanoxylon* (blackwood) was planted in the saturated riparian zone. In the top, southern half of the SMZ, *Eucalyptus globulus* (blue gum) was planted on each mound top and there was no saturated riparian zone. All eucalypt seedlings received 200 g of diammonium phosphate within 2 months of planting, which was split between two spade slits in the soil about 15 cm on opposite sides of the planting position. The overall stocking of the plantation was 1419 trees per ha.

A weather station was installed on the boundary of the two catchments in 2007, which recorded rainfall and several other parameters. Farmer records 2 km south of the site indicate average annual rainfall 1991–2006 was 722 mm (range 501–975 mm). The catchments are in steep terrain (average 17° slope). Soils are c. 3 m deep and derived from interlaid slope deposits of cretaceous syenite and permian mudstone.

A 60° aluminium plate, V-notch weir was installed in each catchment in early 2008 and included water level readings with a capacitance probe every 5 minutes. Water level was converted to flow using a standard equation. Water quality measurements commencing in 2007 provided pre-SMZ data. Changes in water quality as a result of the SMZ (treatment catchment) were determined by comparison with these pre-SMZ measurements, and with the adjacent catchment that did not have an SMZ (reference catchment), and with six other nearby reference catchments.

Nitrogen and other parameters were monitored 3-weekly or less frequently using grab samples. Also included in the grab sample program were 6 other headwater catchments within the Forsters Rivulet catchment that did not have SMZs, and which provided replication of the control catchment. Water in both weirs was automatically sampled every few hours for several days during three storms (Table 1); these samples were measured for concentrations of various forms of nitrogen (particulate total N – retained by a 45 mm filter, and dissolved – filtered – total N, ammonium, nitrate and nitrite) and other water quality parameters. At the two weirs, water level was measured at 5 minute intervals and temperature, electrical conductivity, pH, dissolved oxygen, and turbidity were measured at 15 minute intervals.

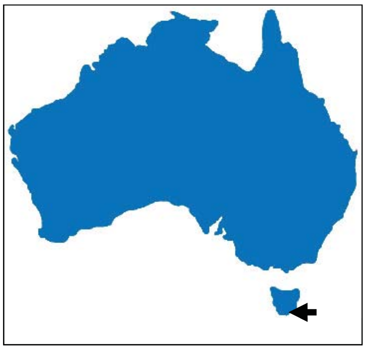


Figure 1. The plantation buffer establishment experiment is located near Cygnet, Tasmania, Australia (indicated by the arrow).

Photo 1. The paired-catchment SMZ experiment consists of catchments with (1) and without (2) an SMZ containing a plantation of eucalypts and acacias planted in August 2008. Additional SMZs shown adjacent (left) and below these headwater catchments are one year older. Water from these headwater streams converge to form Forsters Rivulet, which flows out of the bottom left of the photo.

Table 1. Summary of rainfall and stream flow during the three storms that were automatically sampled during 2009.

	Storm 1	**Storm 2**	**Storm 3**
Rainfall			
Start date	13 May	3 June	26 November
Duration (d)	5	4	5
Total (mm)	33.4	38.4	30.4
Peak Intensity (mm/15 minutes)	1.2	1.4	2.2
Flow			
Average (kL/d)	18.6	117.6	28.5
Peak (kL/d)	35.5	241.4	68.6
Day of peak	3	3	3
Peak:initial	17.6	37.8	279.1

Modelling

The HYDRUS model (Šimůnek *et al.* 2008, version 1.05) was used in a 2-dimensional, sloped, rectangular (trapezoidal) configuration. The model can be accessed at: **http://www.pc-progress.com/en/Default.aspx?hydrus-3d**. Units used were cm for length and mg/L for concentration. An atmospheric (precipitation) boundary condition was usually specified for the surface, with a vertical seepage face at the bottom of the slope, and no-transfer boundaries for other faces. Seepage refers to water movement out of a soil profile at a seepage face with an atmospheric boundary condition (saturation excess), and can include components of interflow soon after rainfall, stored soil water, and ground water entering the soil profile from an aquifer. We use the term runoff to specifically mean overland flow in excess of infiltration. Some authors use the term deep seepage to imply movement of water deep into a soil profile or into an unconfined aquifer. Such a process was not needed in our simulations, but this could potentially be simulated in HYDRUS as a drainage or constant pressure head boundary condition. We used a no-flux lower boundary condition and therefore assumed no interaction with a regional aquifer as a source or sink.

Because we wanted to simulate hillslope processes in two dimensions, an average hillslope length was calculated as catchment area divided by stream length. At least 1,197 spatial nodes were used (96 lateral by 21 vertical). The spacing of lateral and vertical nodes was closest at the lower slope and surface soils zones. Time-steps started at very low values and increased during stable periods to a maximum of 1 d. Simulations were built up by specifying firstly water only, then by adding transpiration (root water uptake), and followed by solute transport. No evaporation rate was included. Before rainfall events were simulated, setting up of a simulation included pre-runs (up to 200 d) of average rainfall and solute inputs that enabled a quasi steady-state to be achieved for seepage rate and concentration. Simulated seepage and runoff fluxes were in two-dimensional units (cm^2/d) and converted to three-dimensional output by multiplying by the length of the third dimension (catchment length = catchment area/length of hillslope = 2 x stream length).

At an early stage, two methods of simulating runoff were tested as follows, i.e. rainfall that is instantaneously in excess of infiltration (method A), and the use in HYDRUS of a hypothetical layer at the top of the soil profile with extremely high porosity and hydraulic conductivity (method B). However, neither of these methods adequately simulated the short-term temporal dynamics of stream flow during rainfall events (Smethurst *et al.* 2009). Instead, a third method was developed whereby measured stream flow was analyzed by the Lyne and Hollick (1979) method to estimate the quick-flow and slow(base)-flow components. The slow-flow component was then routed through HYDRUS as a precipitation input. Resultant seepage estimated by HYDRUS was combined with the quick-flow component in a post-HYDRUS spreadsheet

Photo 2. A view of the farm where streamside plantations are being established in a paired-catchment experiment. Shown in the foreground is the 2008-established buffer on a headwater stream of one of the paired catchments. The plantation buffer consists of Acacia melanoxylon (blackwood) planted in the saturated riparian zone, surrounded by several rows of Eucalyptus globulus (blue gum) or E. nitens (shining gum). In the middle ground is the 2007-established buffer.

to estimate stream flow. Also in the spreadsheet, nitrate concentrations in seepage (as simulated by HYDRUS) and runoff (as user-prescribed values) were combined to provide an estimate of nitrate in stream-flow. In this manner, flow and solute dynamics in the May storm event (storm 1; Table 1) were simulated using inputs summarized in Table 2.

For an annual period that required nitrate transformations (i.e. nitrification and denitrification), and using measured daily rainfall and evapotranspiration, the CW2D module was used with HYDRUS. The CW2D module was designed primarily to simulate nitrate removal from effluent waters draining through flooded constructed wetlands (Langergraber and Šimůnek 2005) by accounting for changes in microbial, organic matter, and some inorganic pools of carbon, nitrogen and phosphorus. The dynamics of 13 solutes (including 3 fractions of organic matter, oxygen and one inert tracer) are simulated using 9 processes and 4 types of microbes. For our application, this complexity was reduced by artificially fixing the depth-dependent concentrations of oxygen and ammonium using hypothetically very high values of the respective solid-liquid phase partition coefficients. The option of including temperature dependency of reactions was not used, and all simulations were conducted using a measured average annual soil temperature (12.5°C). Input setups for specific simulations are summarized in Tables 3 and 4. HYDRUS-CW2D outputs were post-processed in a spreadsheet to estimate annual fluxes and pool changes for water and nitrate.

Table 2. Description of the simulated May storm: salient HYDRUS inputs.

Attribute	Value
General Description	Hillslope as for catchment 1 in photo 1, with tuned water balance. Deep roots (native forest) for top 38% of slope. Shallow roots (pasture) for lower 62% of slope.
Slope length (m)	515.2
Catchment area (ha)	11.15
Duration (d)	7
Water fluxes	Hourly rainfall and potential transpiration assuming no evaporation
Root water uptake	Feddes model parameters (no solute stress): P0 -10, POpt -25, P2H -300, P2L -1000, P3 -1100, r2H 0.5, r2L 0.1
Spatial Nodes	Horizontal: 96 (2.5 m apart at the bottom of slope to 5.7 m apart at the top of slope) Vertical: 21 (0.027 m apart at the top of the soil profile to 0.27 m apart at the bottom of the soil profile)
Time steps (d)	10^{-7}-10^{-3}
HYDRUS units	cm length, mg/L liquid concentration, mg/kg solid concentration, g/cm^3 soil bulk density
Slope (°)	17.4
Rainfall (mm)	33.4
Soil Horizons: thickness (m), texture[1], K_{sat} (cm/d)	Horizon 1: 0.24, sandy loam, $3x10^5$ Horizon 2: 0.23, sandy loam, 106.1 Horizon 3: 2.55, silty clay, 0.48
Root depths native forest: pasture: SMZ (m)	3.0:0.5:0.5

Table 3. Description of annual scenarios: salient HYDRUS inputs.

Attribute	Scenario 1	Scenario 2	Scenario 3	Scenario 4	Scenario 5
General Description	Hillslope as for catchment 1 in photo 1, with tuned water balance, nitrification and nitrate uptake. Model estimates of denitrification and nitrate seepage. Deep roots (native forest) for top 38% of slope. Shallow roots (pasture) for lower 62% of slope.	As for scenario 1, except deep roots (trees) added to bottom 25 m of slope (SMZ).	Hypothetical low slope, high rainfall and nitrate, and higher temperature (18° C). Vegetation as for scenario 1, i.e. no trees in SMZ.	As for scenario 3, except deep roots (trees) added to bottom 25 m of slope (SMZ).	As for scenario 4, plus enhanced carbon supply, less anoxic conditions, and double the width trees in the SMZ (50 m).
Slope length (m)	515.2				
Catchment area (ha)	11.15				
Duration (d)	365				
Water fluxes	Daily rainfall and potential transpiration assuming no evaporation				
Root water uptake	Feddes model parameters (no solute stress): P0 -10, POpt -25, P2H -300, P2L -1000, P3 -1100, r2H 0.5, r2L 0.1				
Spatial Nodes	Horizontal: 96 (2.5 m apart at the bottom of slope to 5.7 m apart at the top of slope) Vertical: 21 (0.027 m apart at the top of the soil profile to 0.27 m apart at the bottom of the soil profile)				
Time steps (d)	0.01-1				
HYDRUS units	cm length, mg/L liquid concentration, mg/kg solid concentration, g/cm^3 soil bulk density				
Slope (°)	17.4		2.0		
Rainfall (mm)	572		1200		
Soil Horizons: thickness (m), texture[1], K_{sat} (cm/d)	Horizon 1: 0.95, sandy loam, 106.1 Horizon 2: 1.41, silty clay loam, 1.68 Horizon 3: 0.64, silty clay, 0.48		Horizon 1: 0.95, sandy loam, 106.1 Horizon 2: 1.41, loamy sand, 350.2 Horizon 3: 0.64, sand, 1000		
Root depths native forest: pasture: SMZ (m)	3.0:0.5:0.5	3.0:0.5:3.0	3.0:0.5:0.5	3.0:0.5:3.0	3.0:0.5:3.0

1. Texture as selected in HYDRUS from default options.

Table 4. Description of annual scenarios: salient CW2D inputs.

Attribute	Scenario 1	Scenario 2	Scenario 3	Scenario 4	Scenario 5
Soil specific parameters (all horizons)	Bulk density 1.5 g/cm^3, Disp L 0.5, Disp T, 0.1, Fract 1, Thlmob 0				
Solute specific parameters (for solutes 1-13[1])	Difus W: 0.072, 0.0456, 0.0456, 0.0456, 0, 0, 0, 0.0801, 0.0801, 0.0801, 0.000801, 0.0000801, 0.05 Difus G: 769, 0, 0, 0, 0, 0, 0, 0, 0, 0, 0, 0, 0				
Oxygen atmospheric boundary condition (mg/L)	11				
Kd for solutes 1-13	10^5, 10^6, 10^6, 10^6, 0, 0, 0, 10^6, 0, 0, 0, 10^7, 0				
Maximum concentrations for uptake (mg/L)	Ammonium 560, nitrate 450, phosphate 1.4				
Temperature	Set constant at 12.5°C, no temperature dependence of reactions				

1 Solutes 1-13 in CW2D are 1 dissolved oxygen, 2 readily biodegradable organic matter, 3 slowly biodegradable organic matter, 4 inert organic matter, 5 heterotrophic organisms, 6 autotrophic *Nitrosomonas*, 7 autotrophic *Nitrobacter*, 8 ammonia, 9 nitrite, 10 nitrate, 11 dinitrogen, 12, phosphate, 13 tracer.

Results

Observations

Various Constituents in Grab Samples

We compared grab samples from the paired catchments and those from the control catchments elsewhere in the Forsters Rivulet catchment for three flow seasons (2007-2009, Fig. 2). Salinity (electrical conductivity, EC) was usually substantially higher in the paired catchments compared to the other control catchments, and the difference was greatest during the relatively dry year of 2008 when the buffered catchment also had consistently higher salinity than its paired control catchment.

Concentrations of *E. coli* were highly variable, and usually no catchment or buffering effect was evident (Fig. 2). However, on two occasions in 2009, which was after the SMZ was established (and hence cattle had been excluded from the stream of that catchment), *E. coli* concentrations were very high in the stream of the control catchment (c. 5600 colony forming units (cfu) per 100 mL) and at the same time concentrations were more than 90% lower in the buffered stream (128-269 cfu/ 100 mL). These samples were taken during the wet, high-flow period of 2009, during which grazing cattle severely disturbed the stream and riparian zone of the control catchment. On other occasions during 2009, *E. coli* concentrations in the control catchment were lower than or similar to those in the buffered catchment.

No patterns in unfiltered total N concentrations were evident (Fig. 2), and although ammonium and nitrate concentrations during the second half of the 2009 flow season were consistently lower in the buffered than in the control catchment, a similar effect was already evident in 2007 prior to SMZ establishment. Hence, lower concentrations of ammonium and nitrate measured in the buffered catchment after SMZ establishment cannot be fully attributed to a buffering effect.

Phosphate concentrations in the buffered catchment were consistently higher than those in the paired control catchment during the 2009 flow season (Fig. 2), and no effect was evident in 2007 or 2008, i.e. prior to the combination of fencing, grazing and high stream flow. This result (and that mirrored in 2009 storm samples – data not presented) strongly suggests that the buffering effect of the SMZ had reduced phosphate delivery to the stream. This effect was not evident in grab samples for particulate or total P or other constituents (DON, pH, dissolved oxygen, and turbidity – data not presented).

Nitrate during storms

Nitrate analyses for storms sampled in May and June 2009 in the buffered catchment indicated initial decreases (dilution) followed by increases (Fig. 3). In November, concentrations overall were much lower than the previous two monitored storms. A dilution phase was evident, but again concentrations increased during the storm. The control (unbuffered) catchment also indicated a concentration increase during the storm in June, but May concentrations were very low (mostly at the detection limit) with a small increase in concentrations evident in two samples during the storm. Nitrate concentrations in the control catchment were high in November with only a minor dilution effect observed and no increases in concentration during the storm.

N Forms

The percentage of N in stream water present as nitrate in both paired catchments ranged from a minimum of 0-2% to a maximum of 40% during the June storm (Fig. 4). Nitrate was general present at a concentration similar to or lower than particulate N (PN), which in-turn was generally less than dissolved organic N (DON). Ammonium and nitrite concentrations usually made up less than 5% of total unfiltered N, but in the control catchment during the June storm, ammonium concentrations almost reached those of PN in three samples. These percentages of N forms cover the range that we generally observed in other storm and grab samples (data not presented) and indicate that DON and PN are the dominant N forms during base flow conditions, and that nitrate (and to lesser extent ammonium) reached similar concentrations to PN during parts of some storms.

From grab samples, we estimated that annual N export during May 2008 to April 2010 was 60-90% dissolved organic N (DON), 12-31% particulate N (PN), and 0-9% nitrate. Total N export during the dry first year was <1kg N/ha in both catchments, and 9 and 6 kg N/ha in the unbuffered and buffered catchments respectively.

High temporal resolution monitoring of turbidity

High resolution temporal patterns of turbidity in control and buffered catchments between April and August 2009 indicated benefits due to the SMZ during both low and high flows (Fig. 5). During the low-flow period 1/4/2009 to 4/6/2009, turbidity in the control catchment was usually more than

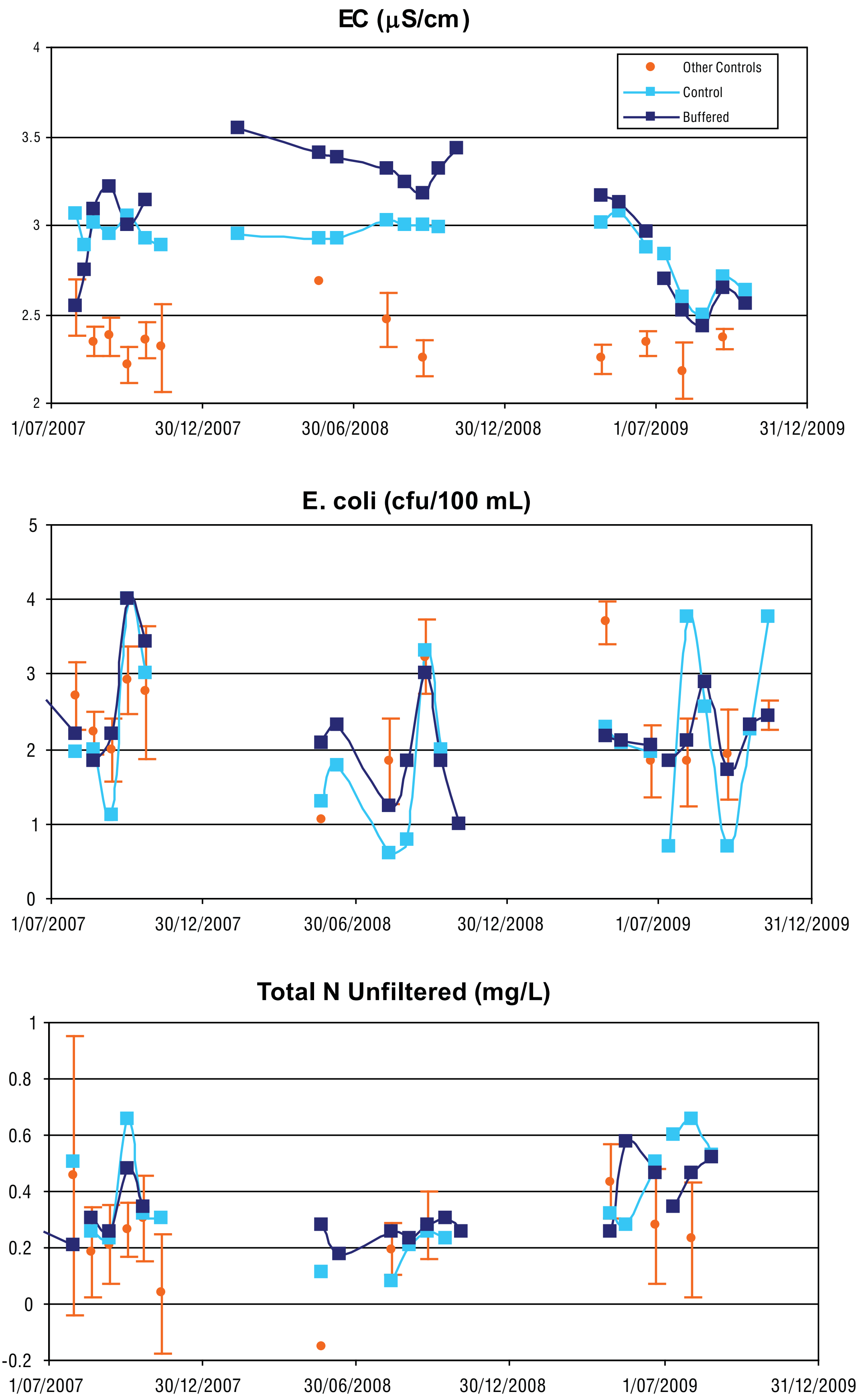


Figure 2. Patterns of various constituents measured during 2007-2009 in water from the paired catchments and headwater control catchments elsewhere in Forsters Rivulet catchment (bars indicate 95% confidence interval of the mean, n = 6). All Y-axis values are on a log_{10} *scale.*

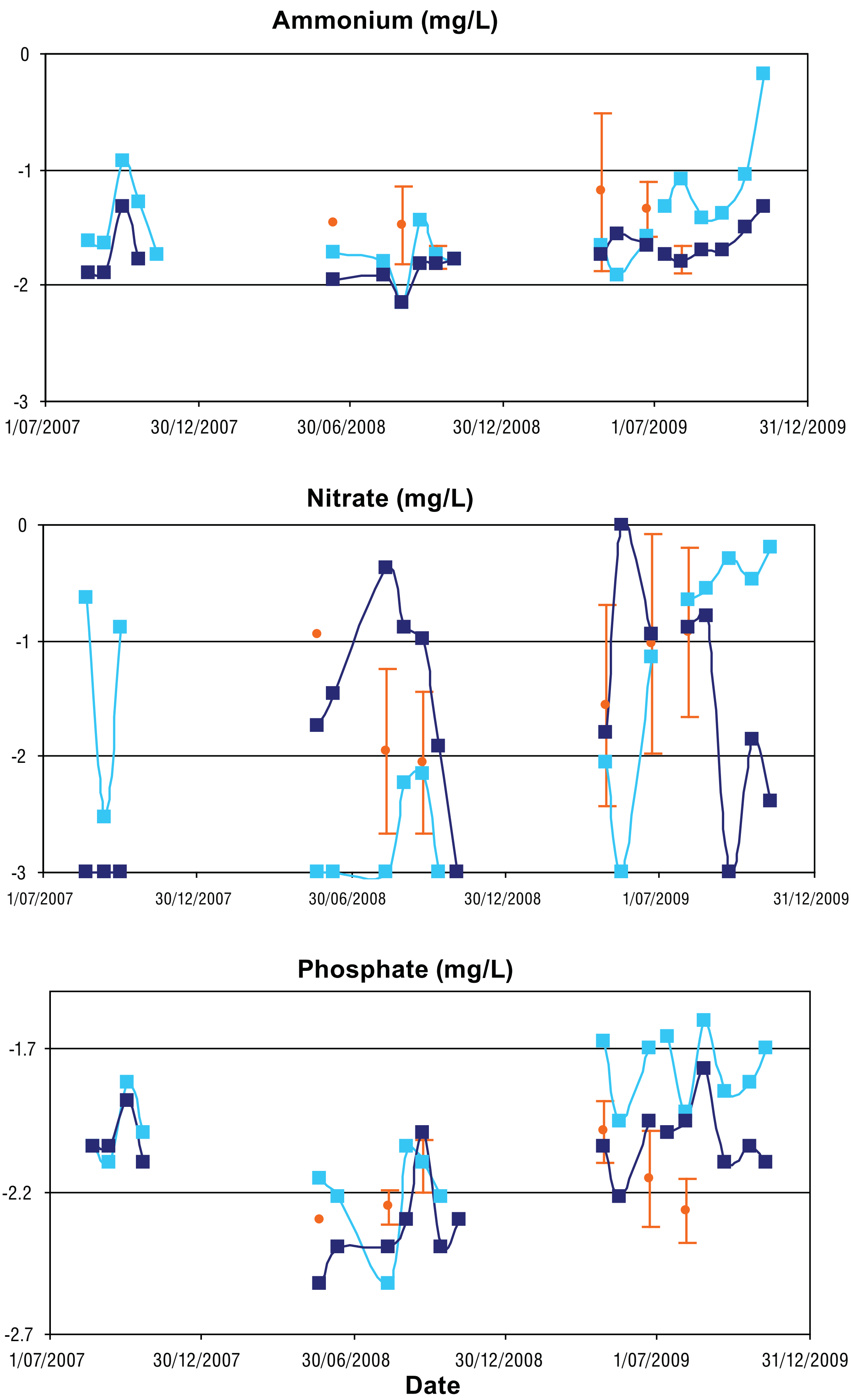


Figure 2 (continued). Patterns of various constituents measured during 2007-2009 in water from the paired catchments and headwater control catchments elsewhere in Forsters Rivulet catchment (bars indicate 95% confidence interval of the mean, n = 6). All Y-axis values are on a $\log_{10}$ scale.

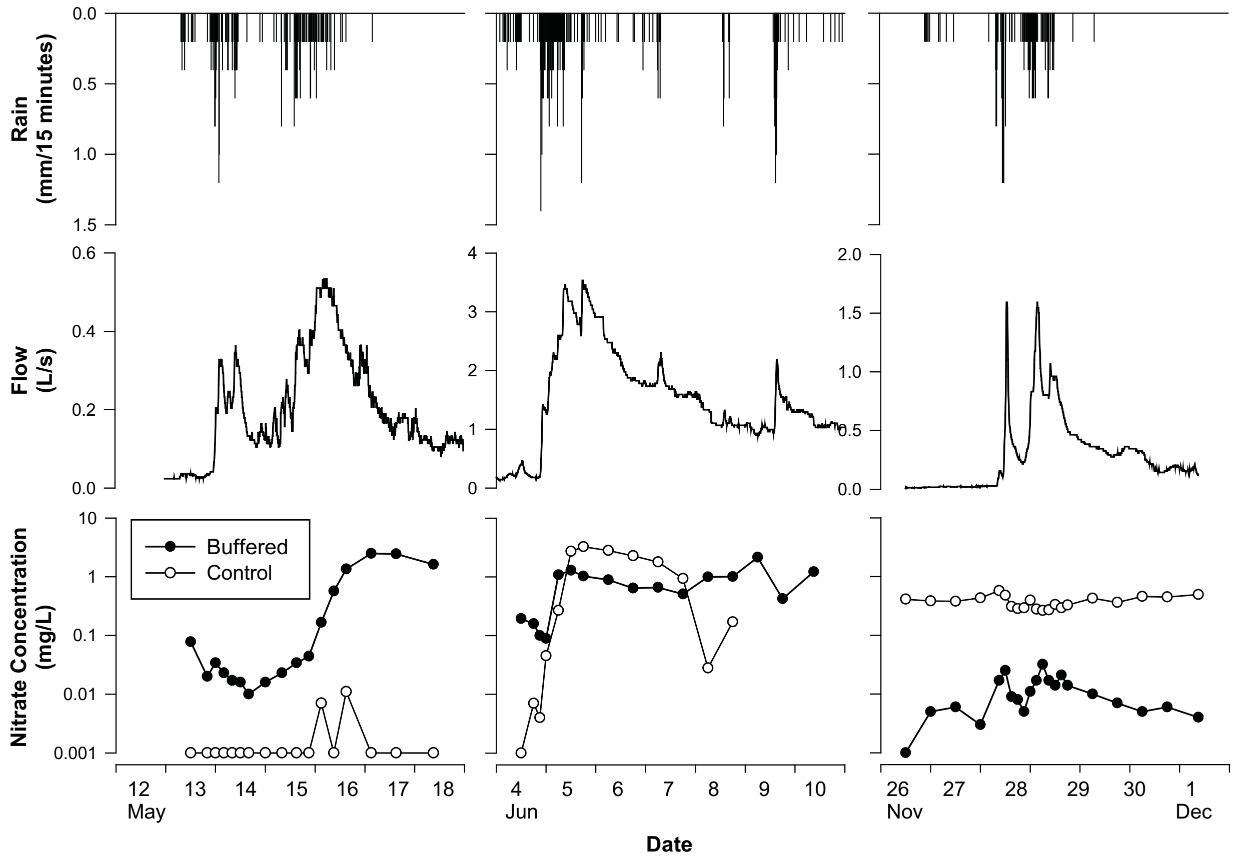


Figure 3. Temporal patterns of rainfall, stream flow and nitrate concentrations during three storms in 2009. Nitrate concentrations are shown for the two paired catchments. Stream flow is only shown for the buffered catchment, but the temporal pattern of relative flow was similar in the control catchment.

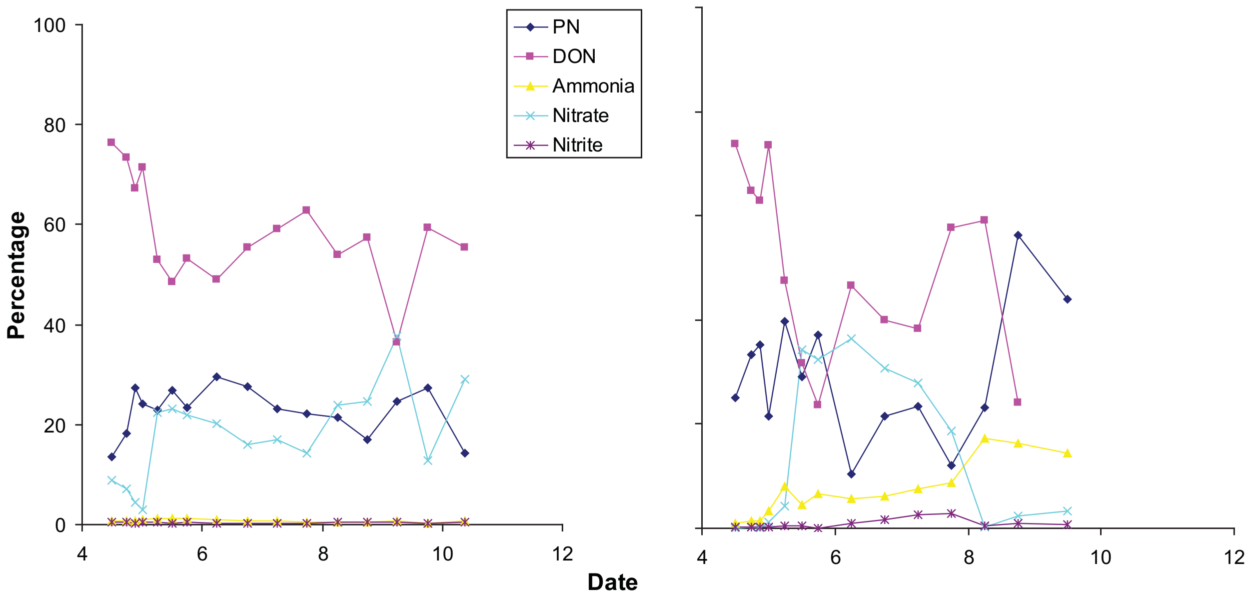


Figure 4. Patterns of percentage contribution of different N forms during the June 2009 storm in the buffered (left) and control (right) catchments.

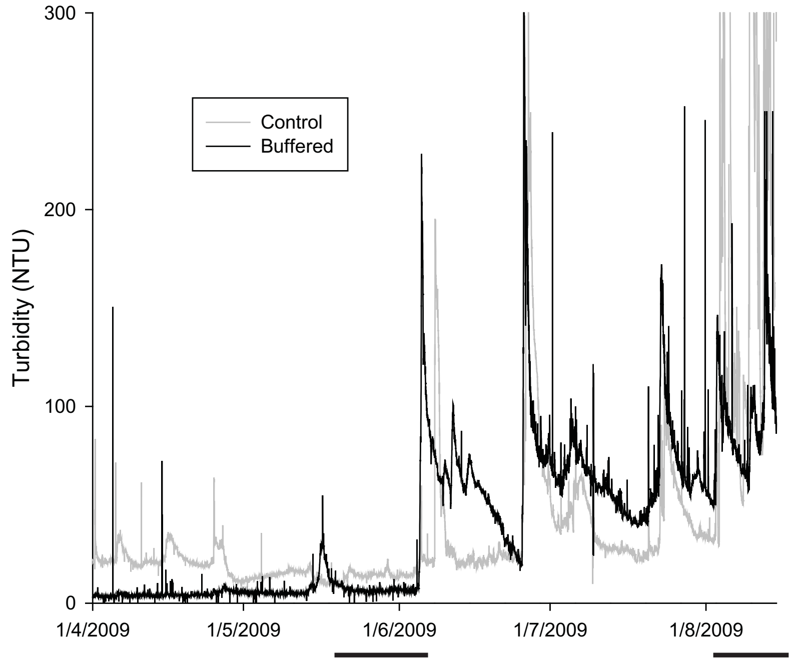


Figure 5. High temporal resolution turbidity patterns in control and buffered catchments between April and August 2009. Cattle were in the paddock during the periods indicated by horizontal bars. During the low-flow period 1/4/2009 to 4/6/2009, turbidity in the control catchment was usually more than twice that in the buffered catchment, except for the 15–19/4/2009, which was during a storm that signalled the autumn break. We attributed higher turbidity in the buffered catchment during these few days to sediment delivery from the cultivated mounds. On the 4/6/2009 tunnel erosion commenced in the buffered catchment and similarly on the 7/6/2009 in the control catchment. Starting on the 3/8/2009 turbidity values in the control catchment were usually much higher than those in the buffered catchment, and often off-scale (>250 NTU maximum for these probes). This period coincided with severe cattle disturbance in the control stream and saturated soil conditions in most of both catchments.

twice that in the buffered catchment, except for the period 15-19/4/2009, which was during a storm that signalled the autumn break. We attributed higher turbidity in the buffered catchment during these few days to sediment delivery from the cultivated mounds, because there was visual evidence of sediment being washed off some mounds, and pits were full of turbid water and had spilled, implying connectivity between mounds and the stream. Cattle presence during this low-flow period in the control catchment appeared to not increase turbidity at that time, but it might have contributed to a cumulative effect detected later.

Peak turbidity values coincided with peak flows (data not shown) and tunnel erosion in both catchments greatly contributed to high turbidity values during the wet winter period. Tunnel erosion, which is a feature of the dispersive soil in these two and similar catchments in southern Tasmania, channelises highly turbid water and therefore bypasses the buffering effect of the SMZ. During peak flows without cattle, turbidity values were equally high in the control and buffered catchments, but between flow peaks turbidity was highest in the buffered catchment. A greater tunnel erosion effect in the SMZ catchment could be due to several effects that we were unable to separate. Firstly, the SMZ catchment size is larger than the control catchment, yet the stream lengths are similar. Hence the water flux per unit stream length, and hence erosive potential, might be highest in the SMZ catchment. Secondly, we suspect that the SMZ catchment is more affected by soil development from syenite rock than occurs in the control catchment, and the saline nature of syenite might be more conducive to dispersive soils than the Permian sediments. Thirdly, some

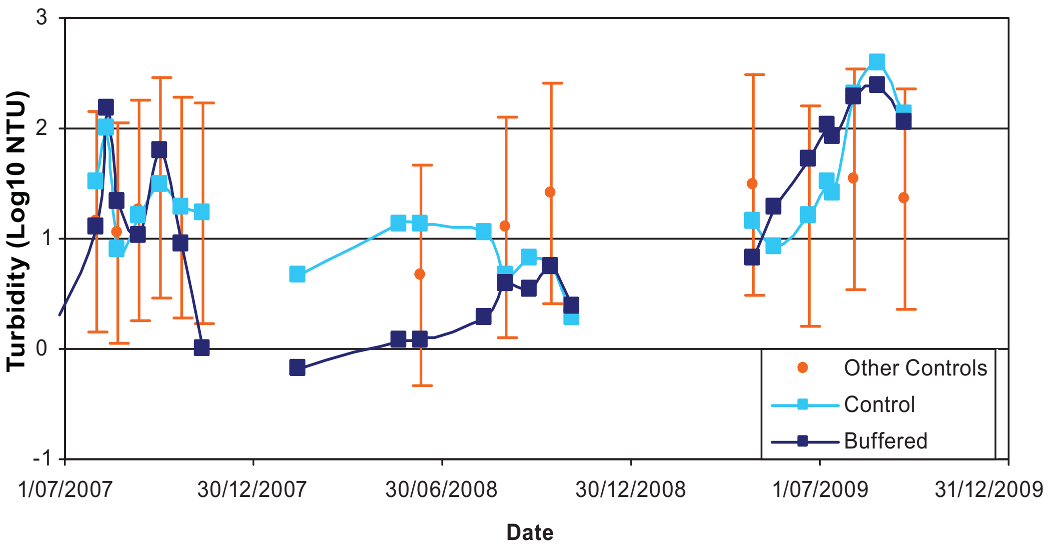
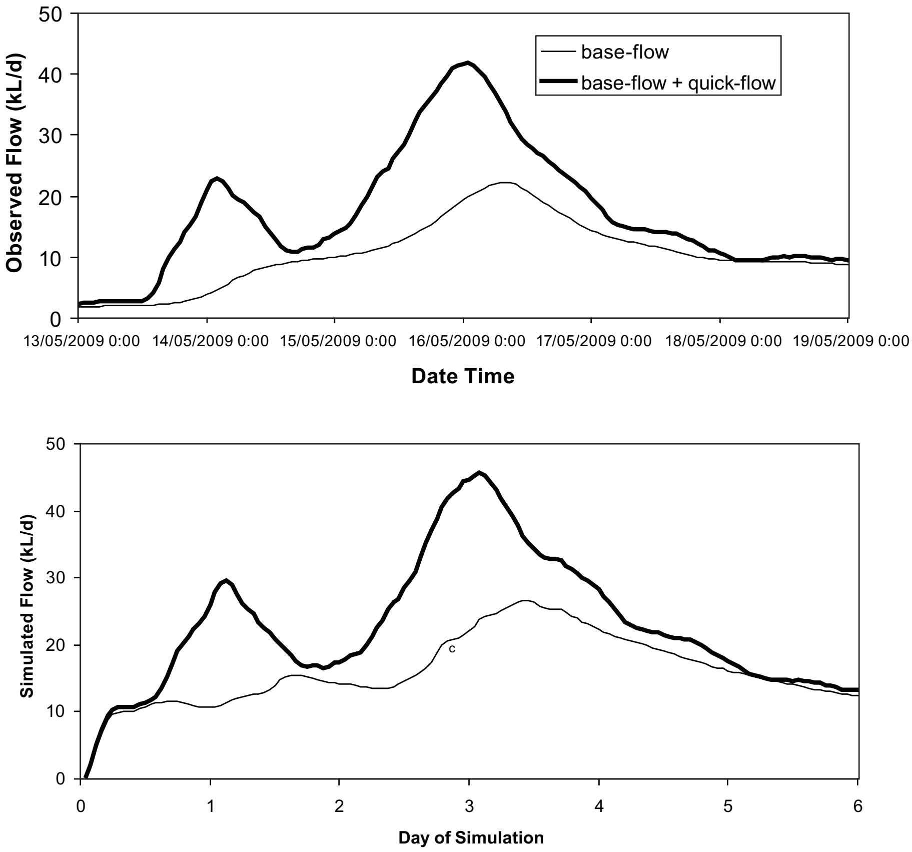
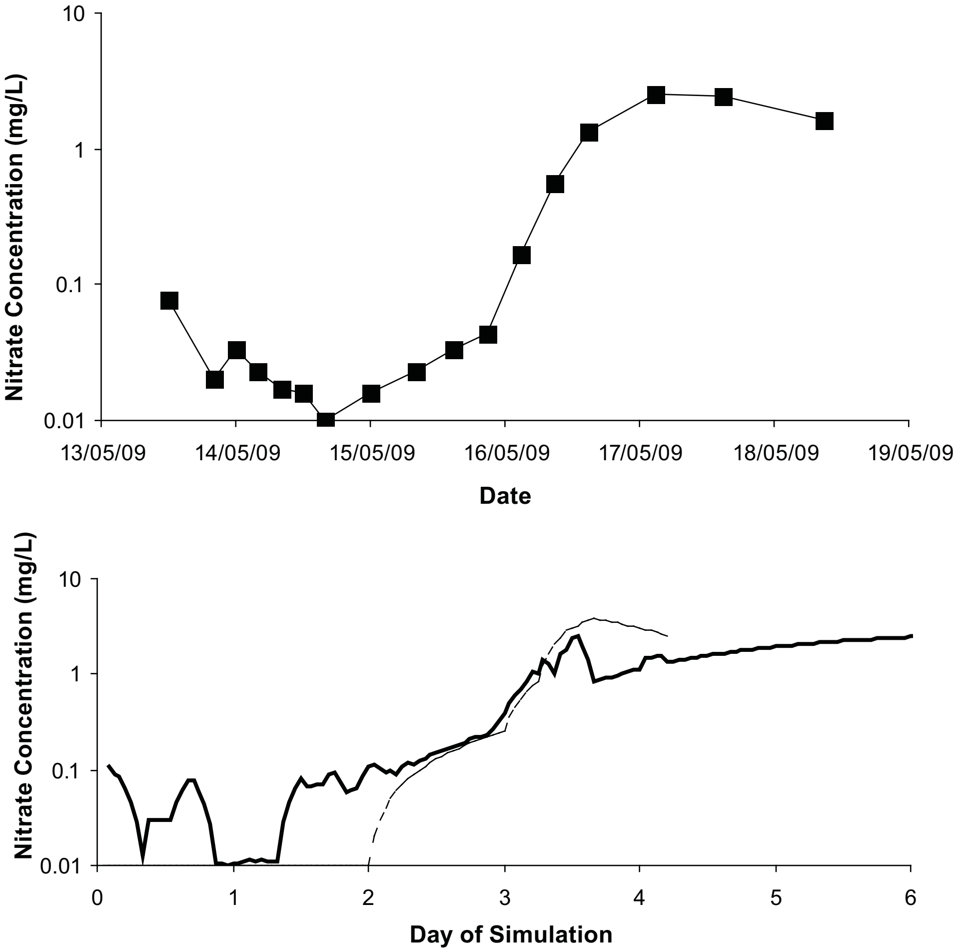


Figure 6. Patterns of turbidity measured during 2007-2009 in water from the paired catchments and headwater control catchments elsewhere in Forsters Rivulet catchment (bars indicate 95% confidence interval of the mean, n = 6). Y-axis values are on a $\log_{10}$ scale.



Figigure 7. Observed flow (top) in the Willow Bend catchment 13-19 May 2009 and its simulation (bottom) using quick-flow analysis and the HYDRUS model.



Figire 8. Measured (top) and simulated (bottom, solid line) concentrations of nitrate-N in stream water at the Willow Bend site. Assumed concentrations of nitrate-N in overland flow (quick flow) are indicated by the broken line in the bottom graph.

cultivation pits of the SMZ intersected pre-existing tunnels in the soil of the SMZ catchment, which might have facilitated or exacerbated sediment delivery in this catchment.

Buffering resulted in reduced turbidity when cattle were present and soils were very wet (Fig. 5). Unfortunately, instrument failure in the control catchment on 16/8/2009 precluded high temporal resolution comparisons after that date. Two grab sampling dates also indicated higher turbidity in the control than in the buffered catchment, and very high variability amongst other control catchments (Fig. 6).

Simulations

May Storm

Routing slow flow through HYDRUS as precipitation and seepage, then recombining seepage with quick flow, reproduced closely the patterns of total observed flow and its estimated slow flow component for the May 2009 storm (Fig. 7). By including high surface soil nitrate and high K_{sat} values (as a substitute for preferential flow), the pattern of nitrate simulated in combined overland flow and subsurface flow (seepage) closely matched the temporal pattern and absolute values of those measured (Fig. 8). However, one source of error is the unknown temporal pattern of nitrate concentrations in overland flow. The potential contribution of nitrate in overland flow can be seen in Fig. 8, where low nitrate (0.01 mg/L) was assumed during the first two days, and peaked at 3.8 mg/L at 3.7 d.

Annual Scenarios

By tuning parameters that controlled soil water status, nitrification and uptake, observations of these components of the buffered catchment were

Table 5. Simulated water and nitrate dynamics at the Willow Bend Farm site for the period May 2008 to April 2009. Setting up of the model required various parameters to be tuned to achieve the required water balance and rates of nitrification and nitrate uptake in the pasture base case (scenario 1). To simulate tree vegetation in the SMZ (scenario 2,) roots were extended to the full depth of the soil profile in the SMZ (lowest 25 m of slope). Tuned values are shaded grey.

Pool or Flux	SMZ Vegetation	
	Pasture (scenario 1)	Trees (scenario 2)
Water Balance (mm)		
Precipitation	572	572
Evaporation	0	0
Transpiration	616	625
Overland flow	15	15
Seepage	3	0
Soil Water	-63	-69
Runoff Coefficient (%)	3	3
Nitrogen Balance (kg ha^{-1} $year^{-1}$)		
Nitrification	154.1	152.6
Uptake	127.3	121.5
Denitrification	2.5	2.5
Seepage	0.0	0.0
Soil nitrate	24.3	28.6
Stream nitrate (mg/L)	0.017	0.000

Table 6. Simulated water and nitrate dynamics at a hypothetical site with lower slope, higher flow and higher nitrate concentrations than at the Willow Bend site. Setting up of the model required various parameters to be tuned to achieve the required water balance and rates of nitrification and nitrate uptake in the pasture base case (scenario 3). To simulate tree vegetation in the SMZ (scenario 4,) roots were extended to the full depth of the soil profile in the SMZ (lowest 25 m of slope). A further scenario (scenario 5) included extended the width of the SMZ to 50 m and reduced the level of anoxia in the water table from 1 to 5 mg O/L. Tuned values are shaded grey.

Pool or Flux	SMZ Vegetation		
	Pasture (scenario 3)	25 m tree buffer (scenario 4)	50 m tree buffer plus less anoxic (scenario 5)
Water Balance (mm)			
Precipitation	1200	1200	1200
Evaporation	0	0	0
Transpiration	811	806	843
Overland flow	170	189	162
Seepage	228	200	173
Soil Water	8	41	31
Runoff Coeff. (%)	32	31	28
Nitrogen Balance (kg ha^{-1} $year^{-1}$)			
Nitrification	701.7	702.6	729.7
Uptake	255.9	247.4	265.9
Denitrification	12.6	12.8	3.6
Seepage	0.0	0.0	0.3
Soil nitrate	433	441	460
Stream nitrate (mg/L)	0.008	0.008	0.081

adequately simulated (Table 5, scenario 1). By adding deep roots to a 25 m SMZ (scenario 2), predicted transpiration was increase by 1% and seepage halted. The runoff coefficient in both cases was 3%, which is indicative of the dry year that was simulated. Uptake was 83% of nitrification, and denitrification was predicted to be only 1.6% of nitrification. There was negligible export of nitrate in stream flow (seepage), and adding deep roots to the SMZ had little affect on nitrate fluxes.

Because rates of denitrification in scenarios 1 and 2 were very low, a hypothetical scenario was developed that was wetter, warmer, of lower slope and of higher nitrate, and that thereby increased rates of denitrification from 2.5 to 12.6 kg/ha/year (Table 6, scenario 3). Adding deep roots to a 25 m SMZ (scenario 4) surprisingly decreased transpiration by 0.6% and seepage by 14%, as overland flow increased by 11% and the overall runoff coefficient decreased from 32% to 31%. There were predictions of only minor changes to nitrate fluxes.

In a further case (scenario 5), transpiration was increased by 5% by doubling the width of the SMZ with deep roots. In addition, we assumed tree roots would lower the water table and lead to more aeration of subsoils, and also add carbon. Nitrification was increased by reducing anoxia from 1 to 5 mg/L dissolved oxygen in the subsoil, and increasing the maximum potential rate of nitrification. Concentrations of readily and slowly available organic matter were doubled and higher potential rates of denitrification used (Figs. 9-10). The balance of these effects was to increase nitrification and nitrate uptake by 4% and decrease denitrification by 71%.

The average nitrate concentration in stream water was calculated assuming overland flow contained no nitrate. Average concentrations therefore depended mostly on the volume of seepage and its nitrate concentration. Results from these scenarios (Table 6) suggest that, if trees reduced anaerobic conditions in the riparian zone and thereby denitrification, average nitrate concentrations in stream water could increase despite reduced seepage and increased water and nitrate uptake by buffer vegetation. This result was not observed at our site (Figs 2 and 3). These results highlight the complexity of predicting the integrated effects of buffering on stream nitrate concentrations.

Discussion

Buffering Effects

The potential buffering effects of SMZs are well established. For example, Zhang *et al.* (2010) summarised data from 73 published studies and concluded that they can be very effective at removing sediment, N, P and pesticides. From incoming water, 97% of the sediment, 93% of pesticides, 92% of N, 90% of P were removed on average with buffers of c. 20 m wide buffers. Because removal efficiency as a function of buffer width was asymptotic, even narrower buffers removed substantial amounts of contaminants on average. A limitation of the dataset was that it was largely limited to plot- or paddock-scale studies, and it was dominated by overland flow measurements that are important for colloidal or sediment-associated contaminants, but less important for some dissolved contaminants. Many

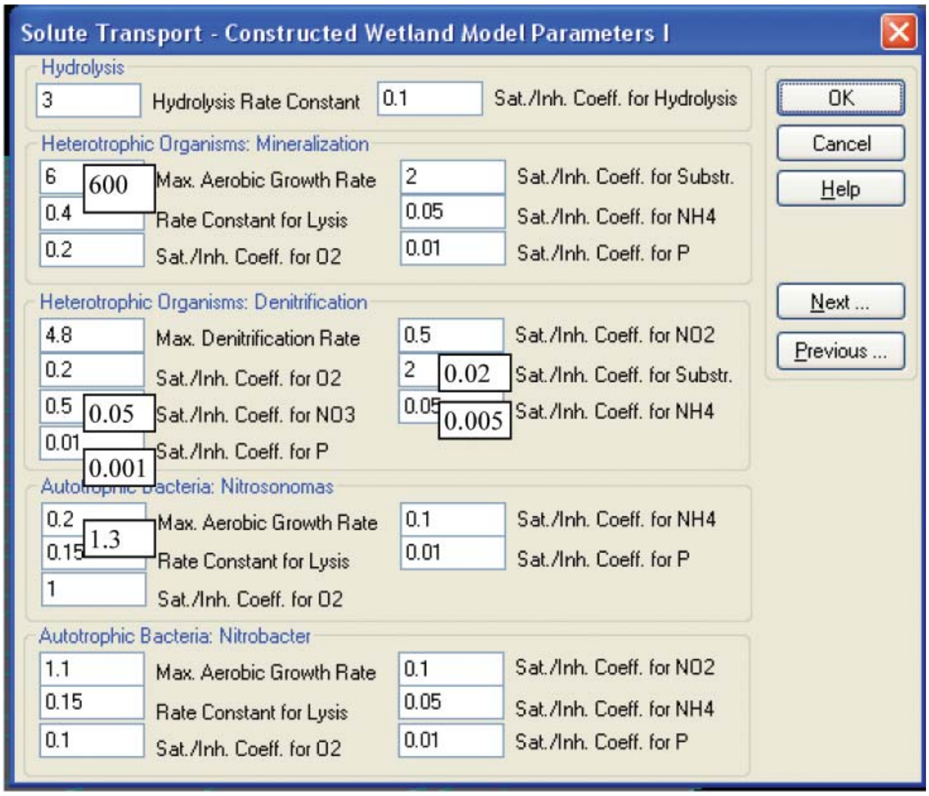


Figure 9. Input parameters for CW2D microbial growth. The second value for some parameters was that used for scenarios 3-5, which increased rates of nitrification and denitrification.

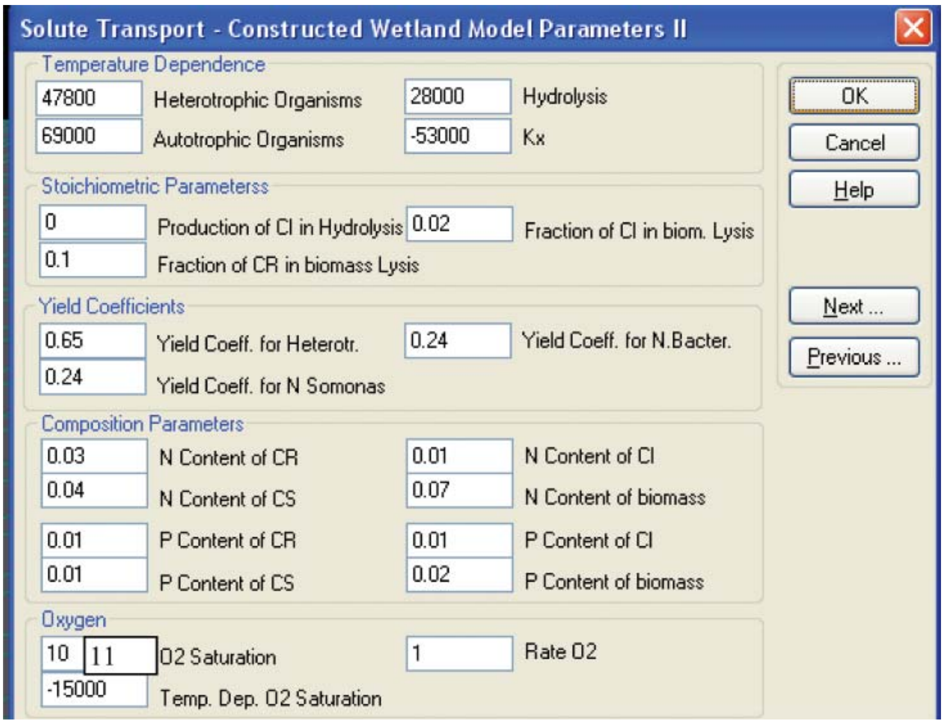


Figure 10. Input parameters for CW2D stoichiometries and reaction rates. The second value for one parameter was that used for scenarios 3-5, which increased the rate of nitrification.

of these studies would not have captured processes that are important at larger scales, e.g. flow concentration effects (Fox *et al.* 2010). More information is also needed on subsurface removal efficiencies for nitrate and phosphate at catchment scales.

In general agreement with Zhang *et al.* (2010), we found that phosphate concentrations and some low- and high-flow turbidity values were substantial reduced by the SMZ, and that this occurred within a year of its establishment. We also observed positive effects on *E. coli* counts on two occasions, but we did not target sampling for *E. coli* to coincide with storm events that would have yielded the highest counts (McKergow *et al.* 2010).

The buffering effects of SMZs often exhibit a lag in response of water quality or populations of desirable organism in the order of years or decades, even when such practices are well-designed and implemented (Meals *et al.* 2010). This lag can be due to delays in the effect being delivered to the water resource, the time required for the water body to respond, and the effectiveness of the monitoring program to detect the response. For example, in one SMZ plantation study it took 8-12 years to achieve substantial reafforestation (Newbold *et al.* 2010).

Although the evidence of Mayer *et al.* (2007), Zhang *et al.* (2010) and the studies cited therein is very strong that buffers can have a strong mitigation effect on water quality, we also need to recognise that in some circumstances a measureable effect on water quality has not eventuated. For example, in the Choptank River Catchment, USA, there was no improvement in stream water N and P concentrations during the period 2003-2006 despite 11% restoration of streamside vegetation during 1998-2005 (Sutton *et al.* 2010). Possible reasons for this lack of effect were insignificant area (width by stream length), connectivity and maturation of the buffers, and increased agricultural inputs. Agricultural drainage networks can also allow contaminated water to bypass vegetative buffer systems. For example, an SMZ containing commercial plantations species in the Bear Creek Catchment, Iowa, USA, was established in a region with large networks of subsoil field drainage systems that provide the majority of base flow to some streams (Shultz *et al.* 2009). Nitrate and phosphate are transported from below the crop root zone directly to the stream bypassing the SMZ.

A concern about using commercial forest plantations in SMZs is that inappropriate harvesting might lead to increased sediment delivery to the streams they were designed to protect. This effect might result from disturbed soil and a reduced sediment filtering effectiveness. This concern was largely allayed by Neary *et al.* (2010) who reported that harvesting SMZs using best management practices largely avoids sediment production. Further, the nutrient sink strength and mitigation capacity of buffers might need to be rejuvenated periodically by removing nutrients contained in plant materials by careful harvesting, grazing or mowing (Dosskey *et al.* 2010).

Modelling

During the past couple of decades, interest has increased in developing models that include dynamic, within-soil processes that govern the transport and composition of water delivered to steams (e.g. Creed and Band 1998). Much effort has focussed only on water, and one cannot hope to successfully simulate solutes if the pools and fluxes of water are not first understood and represented. These modelling efforts have developed at various temporal and spatial scales and using different modelling methods. For example, Chen *et al.* (2010) used the TOPMODEL, the spatial variability of soil properties, and the temporal variability of precipitation and evapo-transpiration to simulate over several years overland- and base-flows in catchments of about 40 km^2. At much smaller spatial and temporal scales, Hilton *et al.* (2008), Guan *et al.* (2010) and Lorentz *et al.* (2008) used the HYDRUS model to simulate runoff from green (grassed) rooves and short sections of hillslopes during storms.

Solutes have also been incorporated in these types of models. Neumann *et al.* (2010) used the Thales model to examine the effect of spatial variability in soil properties on annual delivery of salt, sediment and phosphorus to the catchment outlet. Rassam *et al.* (2008) used the HYDRUS model to suggest where in a catchment the greatest potential rates of denitrification occurred. Krause *et al.* (2009) tested the JAMS/J2000-S model for simulating water quality in a 540 km^2 catchment.

Despite these advances, much complexity is avoided in many models by capturing complex processes in one or a few empirically calibrated factors. The complex interactions of overland flow and seepage processes are manifest in concentration-discharge (C-Q) relationships in the receiving water during storms. Evans and Davies (1998) categorized various C-Q patterns in relation to the dominance of rain, soil or ground water, but until recently these had not been simulated mechanistically.

Haygarth *et al.* (2004) and Holz (2010) also identified several types of concentration-discharge relationships for solutes that depend on chemical form, source and transport mechanism. Haygarth *et al.* (2004) identified that a future challenge was to develop quantitative models that simulated these different situations. Vidon *et al.* (2010) identified a similar need for simultaneously modelling both transport-driven and process-driven phenomena in catchments. Weiler and McDonnell (2006) adapted the Hill-Vi model to demonstrate how complex hillslope water dynamics could be coupled with depth-dependent solute concentrations to conduct virtual experiments for producing C-Q relationships and typical flushing patterns of nitrate, dissolved organic carbon, and dissolved organic carbon. In their simulations, use of the depth-dependent specification of solute concentrations avoided the need to mechanistically simulate these concentrations, which is a much harder challenge.

To provide a more mechanistic method of simulating solute processes within the soil profile in a catchment context, Smethurst *et al.* (2009) demonstrated how C-Q relationships could be generated using the HYDRUS model, but problems remained with simulation of the overland flow component. For this current report we used an alternative method of including overland flow and thereby reproduced the nitrate flushing phenomenon observed in our catchment (Figs. 7-8), and by invoking its nitrogen module (CW2D) we adequately simulated annual water and nitrogen balances (Tables 5-6). This result represents an important development in the application of HYDRUS to hillslope and small catchment situations, because it provides a means of empirically including overland flow and mechanistically simulating within-soil processes.

Whilst applying the HYDRUS-CW2D model to our hillslope situation, we identified various aspects of the model that should be considered for further development (see Conclusions and Recommendations, and Appendix 1). Key amongst these in HYDRUS is the inclusion of overland flow. Overland flow is a focus of model development in its own right because of the complexity and importance of the process for predicting stream flows and erosion. An example is provided by Bhardwaj and Kaushal (2009), who used similar mathematical methods to those in HYDRUS, i.e. Richards equation for water transport and use of a finite element solution method.

Conclusions

1. Monitoring of stream water in the paired catchment experiment during the first full flow season after cattle exclusion and plantation establishment provided strong evidence that the SMZ treatment consistently reduced concentrations of phosphate by up to 70% (0.020 mg/L without the SMZ, 0.006 with the SMZ).
2. On two occasions under these very wet conditions, we observed that spikes in *E. coli* concentrations of c. 5600 cfu/100 mL without the SMZ were mitigated by the SMZ (128-269 cfu/100 mL).
3. Turbidity was also reduced by 30-80% in dry weather conditions (e.g. 20-40 NTU without the SMZ, < 10 with the SMZ) and when cattle were present in very wet conditions (>250 NTU without the SMZ, 150-240 NTU with the SMZ). SMZ establishment led to a small transient increase in turbidity (c. 15 NTU above that in the non-SMZ catchment) during the first major storm of the season, and cultivation might have exacerbated tunnel erosion that is common in similar catchments in southern Tasmania.
4. Patterns of particulate N, dissolved N, total N, ammonium, nitrate, particulate P did not seem to change in response to SMZ establishment, but international experience suggests that more positive effects can be expected in the future as these trees age.
5. The HYDRUS-CW2D model was adapted to simulate the salient processes governing water and nitrate dynamics at a hillslope scale. This involved flow analysis to identify the quick- and slow-flow components of stream flow, and routing of slow-flow through HYDRUS as precipitation.
6. Water and nitrate dynamics could be simulated during storms or over annual periods, if overland flow contributions were already known. Uptake of nitrate appeared to be the dominant nitrate mitigation processes over denitrification. Simulations supported concerns that establishing trees in SMZs could potentially reduce denitrification if it leads to greater aeration of the riparian zone or does not add substantial amounts of carbon in the root zone.
7. Simulations demonstrated the potential usefulness of including mechanistic soil processes in the simulation of catchment hydrogeochemistry. However, much more data on these types of fluxes at a hillslope or small catchment scale are needed to support further model development and validation.
8. For hillslope or headwater catchment simulations, a priority for HYDRUS development is to include overland flow processes and diffusive nutrient supply to uptake surfaces. For CW2D, the priority is to simplify the representation of nitrification and denitrification dynamics, which would also require a more empirical approach and reduce the run-time considerably. An example is provided by the denitrification module of the APSIM suite of models (Thorburn *et al.* 2010). This module also splits nitrogen emissions into N_2 and N_2O forms, which has important greenhouse gas implications.
9. Guidelines are provided for setting up and interpreting hillslope simulations using the HYDRUS-CW2D model. Without CW2D, this method should be suitable for solutes where there is a need to mechanistically simulate the effects of with-in soil processes on concentrations in stream water. The CW2D module provides an example of how modules can be developed for HYDRUS to account for solute dynamic processes that are otherwise not already provided for in HYDRUS.

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Appendix 1. Guide to Using HYDRUS-CW2D for Simulating Catchment Nitrate Dynamics

This guide assumes a working knowledge of HYDRUS version 1.05 and its CW2D module, for which manuals, training, and other forms of support are available online (www.pc-progress.com/en/Default.aspx?hydrus-3d) and periodically in courses and workshops. This guide builds on these resources with the aim of assisting the development of hillslope scenarios for use in a catchment context. Only salient details are provided. Additional details are available in an example provided separately.

Pre-HYDRUS Data Processing

Spreadsheet 1 – Flow Analysis.xls

1. Conduct a flow analysis using, for example, the Lyne and Hollick (1979) method, as available in *Spreadsheet 1 – Flow Analysis*. The key output of this analysis is a time series of slow-flow and quick-flow.

HYDRUS Set-up

Domain Geometry

2. Geometry Information: Within HYDRUS, set up the geometry as a 2D sloped rectangle (rhomboid) to represent your hillslope. Calculate the average slope length as catchment area divided by twice the length of stream(s). Depth should be that of the average soil profile in your catchment. Slope should be the average for your catchment. More complicated geometries are probably possible, e.g. uneven top and bottom surfaces and therefore depths, but we did not test this possibility. At Type of Geometry, choose 2D – Vertical Plane XZ. At Domain Definition choose Rectangular. At Units we used cm.
3. Choose the arrangement of spatial nodes. We spaced nodes c. 2.5–10 m horizontally and 2.75–25 cm vertically, with closer nodes at the bottom of the slope and at the top of the profile. It is important to define early a workable geometry and set of spatial nodes, because some later inputs need to be re-entered when geometry or nodes are changed.

Flow and Transport Parameters

4. Main Processes: When using CW2D, select all options except Inverse Solution, i.e. Water Flow, Solute Transport, Heat Transport, and Root Water Uptake. If you do not wish to use CW2D, e.g. in pre-runs to tune the water balance, select only Water Flow and Root Water Uptake.
5. Time Information: We chose Days as the time Unit, and Time-Variable Boundary Conditions.
6. Water Flow – Soil Hydraulic Model: Preferential flow options are not available when using CW2D. So, matrix flow parameters will need to be adjusted to approximate water dynamics as required. We used the van Genuchten – Mualem model with no hysteresis.
7. Solute Transport: We set mass unit as mg, as concentration is then reported as $\mu g/cm^3$, which equals mg/L. At Reaction Parameters, set any constant boundary condition concentrations required, e.g. dissolved oxygen at cAtm of 11mg/L, and cRoot values (maximum concentration of root uptake) for N and P uptake by massflow. Kd values of each material (soil horizon) can also be set as desired. We used very high values (10^{5-7}) for oxygen, organic matter, ammonium, and phosphate which effectively maintained during simulations the concentrations of the initial conditions anywhere in the soil profile. Constructed wetland parameters are shown in Figs 9–10. Although the Temperature Dependence box had been checked under Solute Information (as instructed for CW2D use), we worked with constant temperature by specifying zero Temperature Dependence of Solute Transport Reaction Parameters, a TBound1 condition of the desired temperature for precipitation inputs, and Temperature Amplitude of zero.
8. Root Water Uptake: We chose to use the Feddes model with No Solute Stress, and grass parameters.
9. Time Variable Boundary Conditions: As precipitation input, copy-and-paste the slow-flow output from the flow analysis (Spreadsheet 1) with the correct units and time steps. Also include Evap. and Transp. as potential rates; we used zero Evap., and Transp. as the potential ET reported by our weather station. Set the temperature of and concentrations in precipitation at TValue2, cVal2–1 (oxygen), cVal2–13. Surface area associated with transpiration: this variable can be tuned to help provide the target water balance.
10. Default Domain Properties: For the top node, set Code = –4 (atmospheric boundary condition). Set the material and root codes for each depth node, as well as the default temperature and liquid and solid phase solute concentrations. When initially setting up a simulation, we usually set pressure head (h) to be as required at the top and bottom of the soil profile, and clicked the

'Linear Interpolation of Pressure Head between the first and final layer'. After long-term simulations with constant average precipitation, a stable distribution of soil water content could be obtained on the hillslope. We found it helpful for model stability to do this prior to introducing variable precipitation. Because CW2D only recognises microbes on the solid phase, default initial values need to be specified here, i.e. for S5–7.

FE-Mesh

11. Rectangular Domain Discretization: Specify the vertical (x) and horizontal (z) node positions. We found it useful to prepare this set of values in an Excel spreadsheet, the copy and paste them to these columns. We set RS1 = 1, and RS2 = 10. Click 'Generate' to generate the mesh. Every time you generate a new mesh, many previously entered input values need to be re-entered.

Domain Properties

12. Default domain properties were set earlier. Here specific nodes can be changed for Material Type and Root Water Uptake. We did not work with Nodal Recharge, Scaling Factors, Anisotropy or Subregions. Observations nodes can be added here for anywhere in the domain.

Initial Conditions

13. Values here for pressure head, temperature and solute concentrations in liquid (L1–4, L8–13) and solid phases (S5–8, S12–13) are initially as set by the default values input earlier. Here values at individual nodes can be changed, e.g. to specify higher organic matter and lower oxygen in the riparian zone.

Boundary Conditions

14. Water flow: use –4 (atmospheric) for the soil surface that will receive precipitation, –2 for the seepage face. The depth of seepage face on the lower vertical side of the sloped rectangle will determine the depth of the maximum saturated zone in the lower slope zone and the profile of soil water content up-slope. We generally kept the lower 50–70% of the vertical face as 'no flux' nodes. A variable seepage-atmospheric boundary condition can be chosen to allow calculation of saturation excess overland flow, which will be necessary if at any time during a simulation if soil at the surface becomes saturated. At those times, precipitation inputs will cease at changed atmospheric nodes and thereby alter water balance calculations. If saturation occurs at the surface without a variable boundary condition, there will be difficulty reaching a mathematical solution within HYDRUS, performance will greatly slow, and even if the simulation proceeds, substantial inaccuracies in the solutions can eventuate. If a variable boundary conditions is needed, set this within 'BDRC Options' as the option 'Apply atmospheric boundary conditions to nonactive seepage face'.
15. Solute transport: set flux nodes as 'Third-type', and vector 2 at atmospheric nodes (to correspond with that vector in the Time Variable Boundary Conditions).

Run-Time

We found that execution of hillslope simulations with CW2D and daily precipitation could be very slow, depending on the occurrence of near-saturation conditions near surface nodes. A typical simulation took 6–7 hours.

Results in HYDRUS

All normal HYDRUS output information is available with CW2D. Output animations can be visualised in HYDRUS for the time layers specified. Default graphical information is also available.

Post-HYDRUS Data Processing

Two spreadsheets to assist with post-HYDRUS data processing are available. *Spreadsheet 2 – Storm Graphs.xls* enables users to collate flow and solute data and to graph their temporal patterns. We used this spreadsheet to graph temporal patterns associated with storm events. *Spreadsheet 3 – Annual Table.xls* enables users to calculate the net fluxes of water and nitrogen that occur during a simulation. We used this spreadsheet to tabulate annual fluxes. Both these spreadsheets draw on HYDRUS output files that are in text format. The files used are *Cum_Q.out* and *solute*.out* (where * is a value 1–4, or 8–13). In these files are tabulated the cumulative values for each time step, from which we calculate rates at each time step (e.g. during a storm) and total fluxes at the final time step (e.g. for an annual period).

Spreadsheet 2 – Storm Graphs.xls

16. Copy all data from file *Cum_Q.out* created by HYDRUS, and paste it into the 'Cum_Q' worksheet such that the top left data values align and therefore that all other data are positioned correctly and provide for accurate calculations. These data will be pasted as text; they then need to be converted to data using the 'Data/Text to Columns' option of Excel.

17. For the solute of interest (e.g. nitrate is solute 10), copy all data from file *solute*.out* created by HYDRUS and paste it into the 'Solute' worksheet such that the top left data values align and therefore that all other data are positioned correctly and provide for accurate calculations. These data will be pasted as text; they then need to be converted to data using the 'Data/Text to Columns' option of Excel.
18. Check in the StormCalc worksheet that all values from the other two worksheets have been correctly 'looked up'. Some filling down of formulas affected by new values might be needed. Copy quickflow values from Spreadsheet 1 – Flow Analysis (correcting for units) across to this StormCalc worksheet into the QFRain column. If there is quickflow, concentrations therein will need to be prescribed in the QFConc column. The maximum number of rows used in each spreadsheet and graph might need to be adjusted.

Spreadsheet 3 – Annual Table.xls

19. Copy the last line of data from the *Cum_Q.out* and *solute*.out* files to the relevant positions in *Spreadsheet 3 – Annual Table.xls* spread, Annual Fluxes worksheet column B rows 91–96. These data will be pasted as text; they then need to be converted to data using the 'Data/Text to Columns' option of Excel.
20. At column T rows 7, 8 and 11, enter manually total precipitation, runoff coefficient and baseflow proportion for your scenario. The respective simulated values are tabulated in column V beside these. Annualised fluxes of water are tabulated in the green shaded area of columns A–K rows 7–28. Average annual flow-weighted concentrations of nitrate are tabulate with yellow shading at Column M rows 26–27 for total flow and slowflow.

Limitations

Despite demonstrating considerable progress, modelling challenges became evident during this research, some of which were specific to HYDRUS–CW2D and others were more generic.

HYDRUS

- HYDRUS had very limited capability to simulate runoff. Only instantaneous, infiltration-excess runoff is reported as a sum for all atmospheric nodes. Overland flow dynamics, include downslope infiltration, surface roughness and slope effects are not modelled. Our flow analysis estimate of quickflow and its bypass of the HYDRUS simulation largely overcame this problem, albeit in a highly empirical manner. Note that an overland flow module for HYDRUS is now available (**http://www.pc-progress.com/en/Default.aspx?h3d-overland**); it simulates water but not yet solutes.
- Precipitation input to individual nodes and the surface as a whole does not account for slope and thereby assumes that the slope horizontal. As such, the precipitation input simulated for high slopes could be seriously overestimated, e.g. a 60o slope would be simulated to have received twice the rainfall as that provided as input, unless this input had previously been cosine-adjusted for slope. This limitation was removed in a later version of HYDRUS (Šimůnek pers. comm.).
- Runoff as saturation excess is possible as a variable seepage zone, but rainfall is not continued on that portion of the slope. Hence, the water balance is altered. Without a variable seepage zone, conditions that approach saturation near the surface dramatically slow or stop the model due to mathematical difficulties.
- Only one root type is specified, wherever they are placed in the simulation domain. However, a two-root-type version of HYDRUS is being tested (Šimůnek pers. comm.).
- Nutrient uptake is simulated only as the mass flow component. This might not be a serious issue for nitrate, for which uptake in many systems is predominantly via massflow. However, for nutrients that are taken up mainly via diffusive processes, e.g. ammonium, phosphorus and potassium, simulated uptake is likely to be greatly underestimated using HYDRUS. A version of HYDRUS is now available that included active and passive compensated uptake (Šimůnek and Hopmans 2009).
- For hillslope applications of HYDRUS, with or without CW2D, it would be very useful to include the pre- and post-processing outlined in the spreadsheets, which would allow easy generation of, for example, graphs like those in Figs 7–8 and tables like those in Tables 5–6.

CW2D

- Over-parameterized for the application reported here, i.e. organic and microbial pools would not be required for simpler, useful soil N dynamics models;
- This contributed to slow performance of HYDRUS-CW2D in our application (7 hours for an annual simulation using daily rainfall), but we recognise that our set-up might not have been optimal for model performance.

- Net changes to the microbial N pool were not included in mineralisation calculations in Spreadsheet 3, because we were only concerned with nitrification. These changes should be included for net N mineralisation calculations.
- Denitrification doesn't split N gasses into N_2 and N_2O.
- We were unable to parameterise O diffusion into the soil in a way that maintained a realistic O profile.
- The preferential flow options of HYDRUS are not available with the CW2D module.
- Nitrification sensitivity to soil water content needs further testing.
- Graphically, nitrate concentrations seemed to be abnormally high in parts of the soil, especially where there was very low soil water content. However, post-HYDRUS calculation of the concentration in seepage water and other aspects of pools and fluxes (Table 5) were realistic and the mass balance check of HYRDUS was acceptable. Hence, calculation of concentrations only for the graphical presentation of HYDRUS-CW2D outputs needs to be checked.
- Also in some graphical outputs some solute fluxes as seepage are incorrectly reported as drainage.

Apart from the above modelling challenges, these sorts of simulations are also data-limited.

- Data were not available for a full definition of the spatial and temporal heterogeneity of the initial and boundary conditions for soil water and nitrate, including preferential flow, saturated and unsaturated hydraulic conductivity, nitrogen turnover and root distribution and activity.
- Water flux associated with overland flow is a highly complex process that itself is difficult to simulate.
- The solute dynamics of overland flow are also highly complex, because they depend on poorly understood and poorly predictable interactions between overland flow and surface soil.
- *In situ* rates of denitrification are extremely difficult to measure in a way that captures spatial and temporal variability.

The implication of data and modelling limitations outlined above is that these types of simulations can only be done as a gross simplification of reality, and that the simplification of using a 2D hillslope model as shown in this research should be satisfactory for many types of applications.