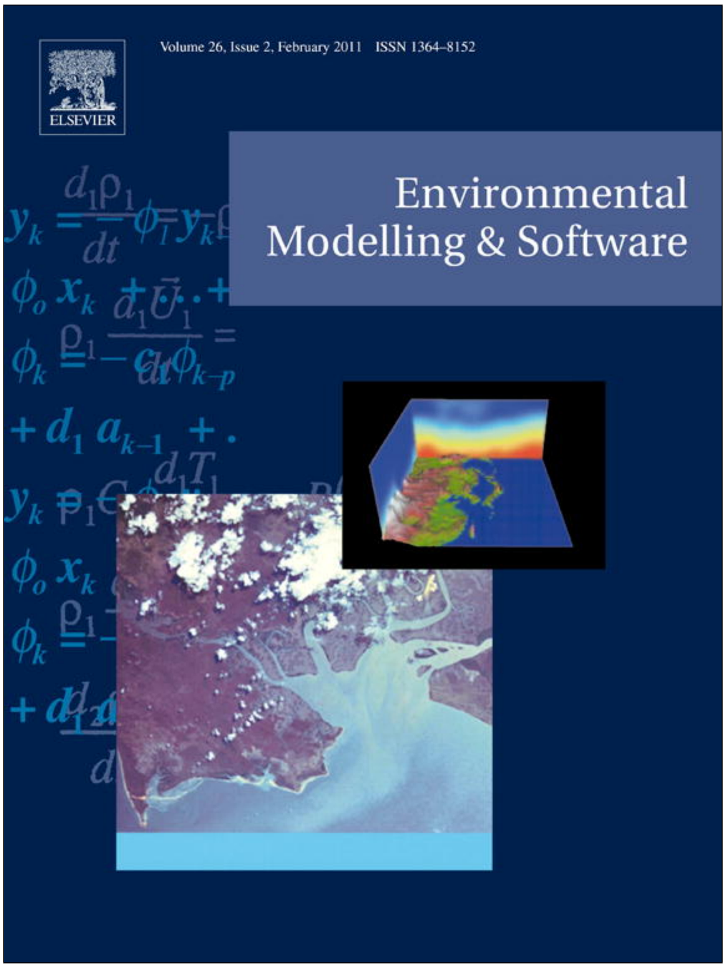
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Integrating farming systems and landscape processes to assess management impacts on suspended sediment loads

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A B S T R A C T

A catchment-scale framework was developed to assess the contribution of sediment sources from farm management actions, gully and streambank erosion on the suspended sediment loads delivered to rivers and associated wetlands and floodplains for two catchments (Avon Richardson, 2885 km^2 and Avoca, 4550 km^2) in Victoria, south-eastern Australia. After considering commonly available data sets, outputs from the point-scale model (HowLeaky2008) were coupled to a catchment scale framework (CatchMODS). Spatially constant, linear scaling factors were used to link point-scale water surplus to streamflow and gross soil loss to hillslope erosion. The model was calibrated against discharge and suspended sediment loads estimated at water quality monitoring gauging stations. Following calibration, estimates of annual and monthly streamflow and 10-year average annual sediment loads were in good agreement with observations. Catchment-scale outputs, particularly sediment loads, were sensitive to scaling factors. The high sensitivity coupled with limited data hindered tight identification of sediment scaling parameters, therefore sediment outputs were uncertain, particularly in the Avoca catchment. Propagation of uncertainty in parameter estimation to model estimates was assessed qualitatively. The boundaries of model estimations were assessed by retaining predictions of behavioural parameter sets, defined as parameter sets that resulted in efficiencies of sediment load and specific sediment yield estimations not more than 5% lower than the efficiency of the optimal parameter set. Under current management conditions, mean annual suspended sediment load at the Avon-Richardson catchment outlet was estimated to be 3350 (3300–3700) t y^{-1}, of which hillslope erosion contributed 65% (60–80%) and gully erosion 35% (20–40%). In the Avoca catchment, annual suspended sediment load was estimated to be 4000 (3500–5100) t y^{-1}, of which hillslope erosion contributed 17% (5–24%), gully erosion 72% (55–93%), and streambank erosion 11% (1–21%). In the Avon-Richardson catchment management scenarios showed that alternative farming systems focussed on retaining vegetation cover throughout the year would yield a 50 per cent reduction of suspended sediment load, estimated at 1700 t y^{-1}. In contrast, fencing and revegetation of connected gullies was estimated to yield the largest reduction in suspended sediment load (1770 t y^{-1}, 44% of current load) in the Avoca catchment. The framework provides an improved tool to make more informed decisions about how much suspended sediment loads can be reduced in response to farm management actions, gully and streambank protection. Its primary strength lies in the ability to propagate farm management impacts to the catchment scale. Other valuable features for use by natural resource management agencies include a high level of transparency, availability of user-friendly interfaces, and a modular structure that provides flexibility and adaptability to new systems.

1. Introduction

As in North America and Europe (National Research Council, 2008; Stoate et al., 2009), water quality problems caused by soil erosion from agricultural systems are becoming increasingly common in Australia (Department of Natural Resources and Environment, 2002a; Summers et al., 1999; Waterhouse et al., 2010). In Victoria for example, in 2002 only 22% of river reaches

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were classified as being in good or excellent condition (Department of Natural Resources and Environment, 2002b). Knowing the contributions from various sources, and costs associated with management strategies for sediment reduction is important for decision-makers such as catchment management agencies and governments to spend available public funding cost-effectively (e.g. Lynam et al., 2010).

Catchment-scale models can be used to assess the contributions of major sources of sediments to a stream network (Drewry et al., 2006). Application of semi-distributed models (i.e. lumped at the subcatchment-scale), such as SedNet (Prosser et al., 2001; Hancock et al., 2007; Wilkinson et al., 2009) have resulted in improved assessment of contributions of hillslope, gully and streambank erosion in medium and large basins (i.e. >3000 km^2; Wilkinson et al., 2009) in Australia, commensurate with the level of limited available water quality data. Although SedNet is one of the most commonly used models in Australia, assessment of the impact of farm management actions (for example changing pasture species or tillage practices) on sediment load reduction is hampered because agricultural land is typically defined into broad land-use categories, and impact of current and alternative land management in specific environmental conditions cannot be quantified.

Point-scale models, which simulates vertical fluxes of water and associated contaminants in a soil column, can be used to quantify the impact of farm management on water quality (sediment, nutrients, and pesticides) of a field or paddock under different soil and climatic conditions (e.g. Rattray et al., 2004; Freebairn et al., 2010; White et al., 2010), and can also be used to assess where farm management is likely to be most efficient in reducing agricultural impacts on water quality (Connolly et al., 1999, 2001; Robinson et al., 2010). The impact of farm management changes on sediment load reduction at the catchment scale, however depends on the environmental context, i.e. catchment hydrology and landscape erosion processes.

Given the importance of gully and streambank erosion processes in many Australian catchments (Australian Government, 2008), coupling point and semi-distributed catchment models has greater potential to improve assessment of farm management impacts at the catchment scale, than the use of uncoupled point-scale models or semi-distributed catchment models alone. Coupling of models at different scales simplifies simulation of complex systems and maintains the process understanding embedded in the single models, but it comes at the cost of higher data requirements, which may not always be available (Hansen and Jones, 2000; Freni et al., 2009; Cerco et al., 2010).

The aim of the research was to develop a catchment scale framework suitable for management decision makers to assess the contributions of farm management actions, gully stabilisation and streambank protection on suspended sediment loads to rivers and associated wetlands and floodplains.

2. Study areas

The framework was developed and tested in two locations, the Avon–Richardson and the Avoca catchments in north-central Victoria, southeast Australia (Fig. 1).

2.1. Avon-Richardson catchment

The Avon-Richardson catchment (36.5°S 143°E) was initially selected for model development and testing. It is an endorheic basin that extends over 2885 km^2 (Fig. 2). Topography is predominantly flat, with over 70% of the catchment having slopes <5%. Rainfall is winter dominant and highly variable; both the Avon and Richardson rivers are ephemeral and can have no streamflow particularly during the summer months (Vigiak et al., 2009).

Agriculture is the dominant land use, with open forest comprising <5% of the catchment area. There are three main farming system zones: grazed pastures in the southern hills (average rainfall approximately 550 mm y^{-1}), mixed farming systems (comprising both grazing and cropping) in the mid-catchment (average rainfall approximately 450 mm y^{-1}), and crop dominant systems in the flat northern part of the catchment (average rainfall approximately 400 mm y^{-1}). Cropping and grazing

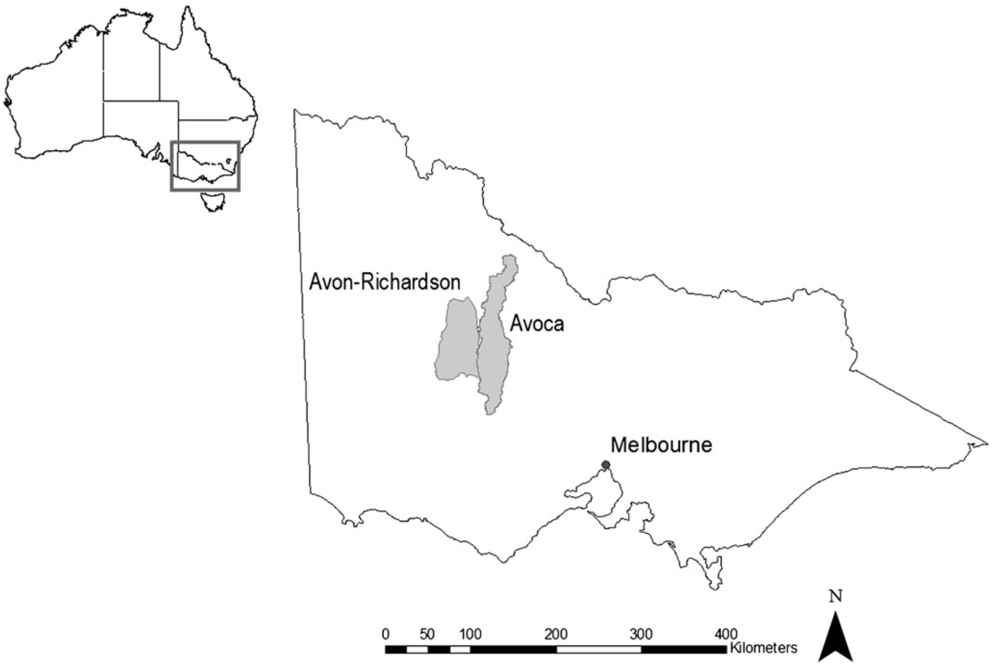

Fig. 1. Location of the Avon-Richardson and Avoca catchments in Victoria (highlighted in the grey box), southeast Australia.

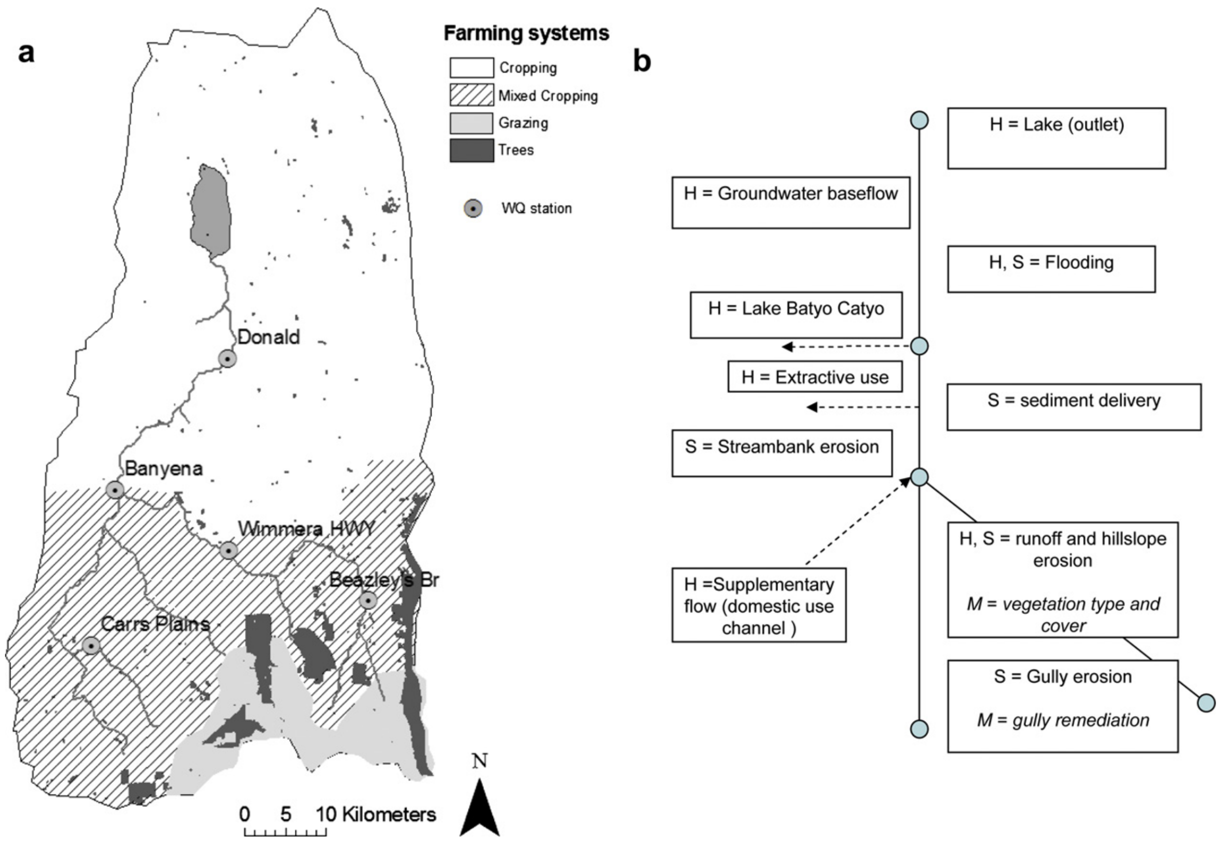

Fig. 2. The Avon-Richardson catchment: a) farming systems, stream network and location of water quality stations; b) conceptual diagram of water and sediment processes: H = hydrology; S = sediment; M = management. Processes that were included in the framework are on the right of the main river line; processes that were recognized but excluded from the framework are on the left.

systems are based on annual plant species and with sheep as the livestock component, similar to many areas of southeast Australia.

The catchment has poor water quality, the 75th percentile of in-situ turbidity measurements being 120 NTU at Wimmera Hwy station, 42 at Banyena, and 20 at Donald (DSE, 2010), whereas the State Environment Protection Policy SEPP Waters of Victoria 2003 sets an acceptable threshold at 10 NTU (DSE, 2005). Hillslope and gully erosion have been identified as the major sources of sediment (SKM, 2003; Australian Government, 2008).

2.2. Avoca catchment

The Avoca catchment (36.5°S 143°E, Fig. 3) is east of, and adjacent to, the Avon-Richardson catchment. The catchment is 4550 km^2 in size, extending from the Pyrenees Ranges and the Black Range in the south, through the Avoca Marshes and ultimately reaching the Murray River in wet years in the north. Slopes are generally of low relief, however hills of the Pyrenees Range and Great Dividing Range in the south of the catchment are steep (Lorimer and Rowan, 1982). Rainfall is winter dominant, ranging from 350 mm y^{-1} in the north to approximately 600 mm y^{-1} in the south.

Farming systems are similar to the Avon-Richardson catchment, predominantly mixed farming in the south and broad-acre cropping in the north. A small but economically important proportion of land in the south (3% of the catchment) is used for grape production and blue gum plantations.

Poor water quality poses concerns for the river health (Anonymous, 2005) with turbidity above SEPP thresholds at all stations but at Amphitheatre, with the 75th percentile of 31 NTU at the Coonooer station and 47 NTU at Quambatook (DSE, 2010). Hillslope erosion, gully erosion and streambank erosion are all important contributors of sediments (Lorimer and Rowan, 1982; Rutherfurd and Smith, 1992; Australian Government, 2008), but there is limited understanding of the relative contributions from each source.

3. Modelling development

The development of the framework was broadly based on the steps of good modelling practice identified by Jakeman et al. (2006).

3.1. Model purpose

The purpose of the research was to provide guidance for water quality management in areas characterized by limited environmental data. The model also helped (i) build understanding of the biophysical systems; (ii) provided a transparent framework to facilitate management discussion; and (iii) allowed 'what if' scenario analysis.

3.2. Modelling scope

Decisions about public investment to improve water quality outcomes are most often made by catchment management agencies and governments. Limited water quality data is a typical constraint. Catchment managers and other stakeholders were involved in an advisory and information seeking role at various

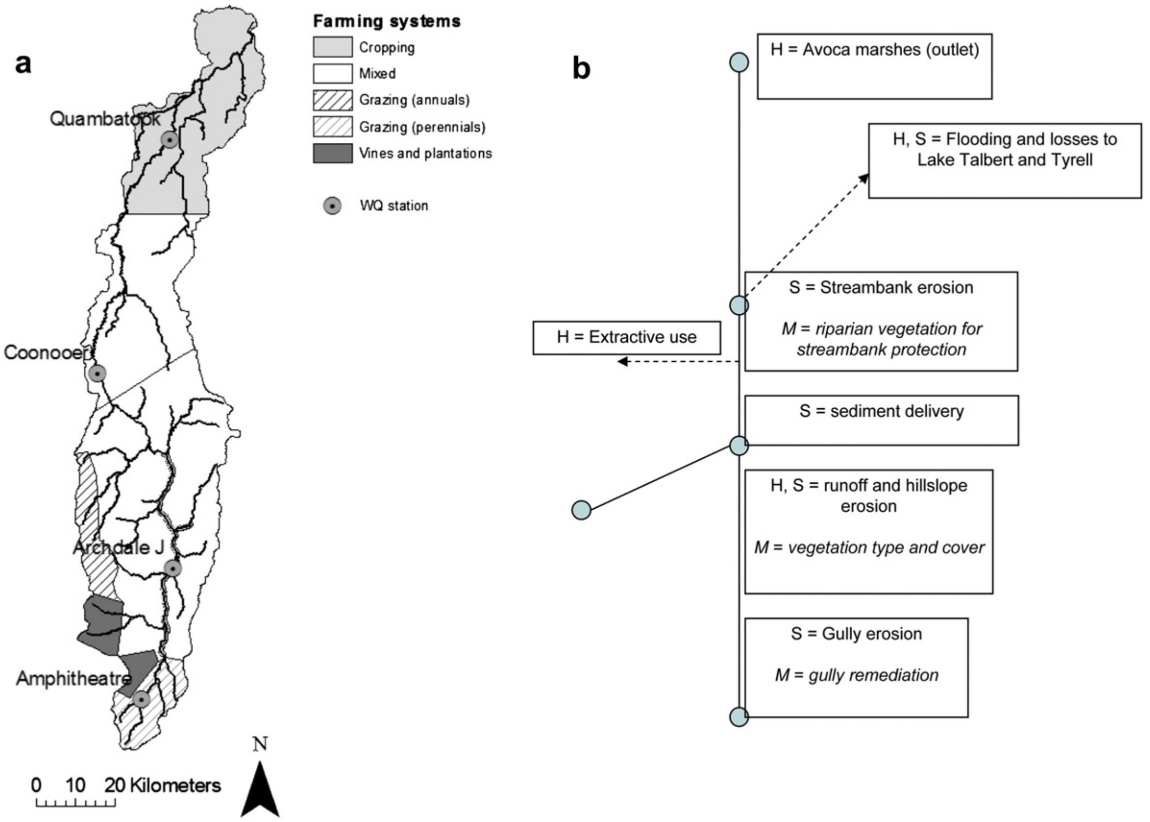

Fig. 3. The Avoca catchment in North Central Victoria: a) farming systems, stream network and location of water quality stations; b) conceptual diagram of water and sediment processes: H = hydrology; S = sediment; M = management. Processes that were included in the framework are on the right of the main river line; processes that were recognized but excluded from the framework are on the left.

stages of the framework's development to maximise the chances of developing a useful decision-making tool.

The scope of the project was to develop and test such a framework which could (i) quantify erosion from gully, streambank and hillslope processes to rivers at a sufficiently fine resolution (spatial and temporal) to enable spatial targeting of management; (ii) assess the impacts of managing sediment via gully, streambank or farm management remediation (preferably in relatively user-friendly, visual way); and (iii) be commensurate with the limited available data.

Although cost of management actions is an important consideration for public investment, we focussed primarily on the biophysical aspects, and excluded detailed economic analysis of management actions. It was recognized that this could be tackled external to the modelling framework, or could be embedded at a later stage. This is a subject for future research.

3.3. Conceptualization of major processes

A crucial phase in the modelling development was conceptualising the dominant processes affecting water quality. The scope of the conceptualisation was initially kept as holistic as possible, with decisions made later on the need for simplification commensurate with data availability and the desired level of model complexity.

Catchment conceptualisation was participative where modellers, stakeholders, and local experts spent a few days together visiting the catchment and discussing major processes affecting water quality and potential management options. Prior knowledge (local investigations and relevant reports from south-eastern Australian catchments) was reviewed. From this, a conceptual diagram of the major catchment processes was drawn and circulated among participants for feedback.

3.4. Selection of point and catchment scale models

The research scope meant that both point scale (representing a field) and catchment scales had to be considered explicitly. The modelling approach pursued was to couple two pre-existing models, a point-scale model and a catchment-scale model. Criteria which guided the selection of models were (i) the system quality, namely robust process conceptualisations commensurate to data availability (Volk et al., 2009; Freebairn et al., 2010); (ii) suitability for stakeholder engagement (Jakeman et al., 2006) including features such as simplicity and transparency of model approach, and availability of user-friendly interfaces; (iii) flexibility of the approach (e.g. Argent et al., 2009) to allow for adaptation to new case studies; and (iv) model developers' commitment to participate and provide long-term support for end-users.

3.4.1. Point scale model

The HowLeaky2008 model (Rattray et al., 2004; McClymont et al., 2008; http://www.apsim.info/How/HowLeaky/howleaky.htm) was selected to simulate land management effects on point-scale water and soil losses. HowLeaky2008 is a 1-dimensional (soil column), daily time step soil water balance model designed to assess the impact of different land management choices on water balance, soil erosion and water quality. HowLeaky2008 comes with an intuitive, user-friendly interface that facilitates interactions with local experts, extension officers and farmers (Freebairn et al., 2003).

The model uses the well-established water balance and erosion components of PERFECT v3 (Littleboy et al., 1999), with modifications in the calculation of deep drainage and soil evaporation (Robinson et al., 2010). The soil water balance is simulated using a cascading bucket approach with up to five soil layers. Runoff is calculated using a modified USDA Curve Number (CN) method, but CN values are adjusted for ground cover (Robinson et al., 2010). Soil evaporation follows the Ritchie (1972) two stage model, and depends on crop cover, soil characteristics and soil water content. Transpiration depends on pan evaporation and crop development (green cover, residue cover, and root depth). Water content above the field capacity of a given soil layer percolates to the layer below at rate equal to or lower than a maximum percolation daily rate (K_{day}, mm day^{-1}). Water that percolates below the deepest soil layer is considered to be deep drainage.

Soil loss is calculated according to (Littleboy et al., 1999):

$$E = F_{cover} \cdot LS \cdot K \cdot P \frac{Q_R}{10} SDR \tag{1}$$

where E is the soil loss (t ha^{-1}); F_{cover} is a function of vegetation cover; LS, K and P are the topography, soil and protection factors of the Revised Universal Soil Loss Equation (RUSLE, Renard et al., 1996); Q_R is the volume of runoff (mm), and SDR is the sediment delivery ratio. Eq. (1) differs from the RUSLE approach in that runoff and not rainfall is the erosive agent, and that the vegetation factor is an explicit function of total vegetation cover. Furthermore, soil loss, runoff and vegetation conditions are calculated for any erosive event (daily output).

HowLeaky2008 can simulate plant growth dynamically (if plant phenologic rules are known) or statically whereby crop growth throughout the year is user-defined. Although the static version may not account well for the impact of annual climatic variability on crop growth, it provides a more stable platform for extrapolation at the catchment scale and allows simulation of land uses for which phenologic rules are not available. The static version was therefore used to simulate crop rotations, pastures and tree cover, having been tested for Victorian conditions previously (Melland et al., 2010). HowLeaky2008 soil loss simulations in the Avon-Richardson catchment were comparable to long-term ^{137}Cs estimates of erosion rates for nearby sites and to erosion rates measured in New South Wales (Vigiak et al., 2010).

3.4.2. Catchment scale model

The Catchment Scale Management of Distributed Sources Model (CatchMODS; Newham et al., 2004; http://icam.anu.edu.au/products/catchmods.html) was selected as the catchment-scale framework. CatchMODS integrates hydrologic, sediment and nutrient export models, and has been previously applied to several catchments (Newham et al., 2004; Croke et al., 2007; Fu et al., 2009; Volk et al., 2009). CatchMODS conceptualizes a catchment into river reaches and associated subcatchment areas (average size of 20–40 km^2), that are linked into a node-link system (Newham et al., 2004). The topology of the stream network enables downstream routing of pollutants with submodels simulating pollutant generation, attenuation, and deposition. The focus of CatchMODS is on the simulation of the suspended sediment fraction, reflecting the importance of suspended sediments as a source and transport medium for many common stream pollutants. The sediment submodel of CatchMODS is modified from the SedNet model (Prosser et al., 2001) retaining several of SedNet's algorithms to estimate hillslope, gully and streambank erosion but adopting a rainfall-runoff model to simulate hydrologic key variables (Newham et al., 2004). CatchMODS was further modified in this framework as described in Section 3.5.

3.5. Model integration and modifications

3.5.1. Spatio-temporal domain

The coupling of the two models required clear identification of spatial and temporal model units at the point and catchment scales (Cerco et al., 2010).

At the point-scale simulation units were called hydrologic response units (HRUs), namely landscape units whose hydrologic behaviour was considered homogeneous. HRUs were defined as unique combinations of climate, soil type and land management. The definition of HRUs was independent of location, and topography was not considered at this scale. Simulations were conducted at the daily time step.

At the catchment scale, HRUs and topography were integrated at the subcatchment level. The geographic properties of each subcatchment comprised a climate zone, the combination of relevant soil types derived from soil mapping, and a topography factor LS. Conversely, subcatchment land management was selected by the user, who allocated fractions of the subcatchment to potential land management options. The fraction of any HRU in a particular subcatchment was calculated as the product of the subcatchment allocated to a land management option multiplied by the fraction of the HRU's soil type. In other words, land management options selected for each subcatchment were proportionally allocated to all soil types occurring in the subcatchment. The approach was in line with the level of detail of available data and enabled analysis of land management scenarios without the need of a Geographic Information System to filter land management and soil combinations.

Hydrologic simulation at the catchment scale was conducted at daily time steps. For suspended sediment budgets, daily point-scale soil losses were aggregated to annual totals. Gully and streambank erosion sediment loads were based on long-term annual average rates. Model outputs were therefore set as 10-year mean annual sediment load. Once again, the level of detail in the output time scale was commensurate to the purpose of the model, i.e. evaluating the long-term impact of catchment management interventions, and data availability. Temporal aggregation of HowLeaky2008 outputs (from daily to long-term annual averages) degraded some of the detail in the dynamics of the system embedded in the point-scale model, but reduced the impact of sources of uncertainty (e.g. extrapolation errors in climate time series; timing of tillage operations in relation to rainfall events) in the model data inputs to the framework output.

Model coupling was one-way and off-line (Cerco et al., 2010), i.e. HowLeaky2008 model outputs were incorporated once into CatchMODS framework before running CatchMODS.

3.5.2. Model modifications

Both models needed modifications to allow their coupling.

The static crop growth module of HowLeaky2008 was modified to allow for simulation of crop rotations, i.e. crop growth sequences longer than one year. The interface was provided with a crop growth editor in which canopy cover, ground cover, and root depth were set at given dates, from which cover and root depth curves were linearly interpolated to any day of the rotation. Eq. (1) was modified to remove the influence of topography on soil loss, i.e. the LS factor and SDR were kept equal to 1.

Several modules of CatchMODS were modified relative to Newham et al. (2004). The hydrologic module was modified in that daily discharge was calculated as a fraction of the total water surplus (i.e. runoff plus deep drainage) generated by the subcatchment HRUs as estimated by HowLeaky2008:

$$U_j = fA_j \sum_{i=1}^{n} \left((Q_{R,i} + Q_{D,i}) \frac{A_i}{A_j} \right) + U_{UP,j} \tag{2}$$

where U_j is the daily discharge (ML day^{-1}) of the jth subcatchment; f is the fraction (0–1) of water surplus that contributes to streamflow (as opposed to the water surplus that drains to deeper aquifers); A_j is the subcatchment area (ha), $Q_{R,i}$ and $Q_{D,i}$ are the runoff and deep drainage of the ith HRU occurring in the subcatchment, A_i is the ith HRU area (ha), and $U_{UP,j}$ is the streamflow received from upstream reaches. Daily discharge was separated into quick flow and slow flow, and routed along the stream using the linear module of the IHACRES rainfall-runoff model (Croke and Jakeman, 2004) to calculate daily streamflow (Q_j, ML day^{-1}). Daily streamflow was used to derive flow-related statistics: mean annual flow ($Q_{MAF,j}$, ML y^{-1}), bankfull discharge ($Q_{BF,j}$, ML day^{-1}, estimated as the daily streamflow corresponding to a recurrence interval t), and the overbank streamflow ($Q_{OB,j}$, ML, the median annual overbank streamflow volume), which regulates flood volumes.

The hillslope sediment module was substantially changed. In previous CatchMODS versions, hillslope erosion was calculated using a basic application of the RUSLE method. In our approach, subcatchment hillslope suspended sediment load (H_j, t y^{-1}) was calculated from HowLeaky2008 HRU's annual average soil loss as:

$$H_j = SDR \cdot LS_j \cdot A_j \sum_{i=1}^{n} \left(E_i \frac{A_i}{A_j} \right) \tag{3}$$

where SDR is the catchment sediment delivery ratio, LS_j is the average RUSLE topography factor of the jth subcatchment, and E_i is the HowLeaky2008 annual average soil loss (t ha^{-1} y^{-1}) of the ith HRU calculated as the sum of daily outputs of Eq. (1) for the simulation period divided by the number of years of the simulation.

The gully erosion module assumed sediments to be sourced from sidewall retreat of permanent gullies. The mass of suspended sediment derived from gully erosion G_j for the jth subcatchment (t y^{-1}) was estimated as (Whitford et al., 2010):

$$G_j = \sum_s L_{s,j} r_s \rho \Delta \tag{4}$$

where for gullies of severity class s, $L_{s,j}$ is the effective length of gullies (m); r_s is the annual average sidewall erosion rate (expressed as annual increase in gully cross section, m^2 y^{-1}); ρ is the bulk density of eroded sediments (t m^{-3}), and Δ is the proportion of suspended sediment in the eroded gully material (i.e. particles <63 µm; Rustomji, 2006). Eq. (4) is similar to the SedNet approach in that erosion is assumed to result from gully walls and is estimated for long-term averages. However, the erosion rates r_s were estimated assuming an exponential decay in the rate of gully erosion during the stabilisation phase in which most southeast Australian gully systems are found (Prosser and Winchester, 1996; Wasson et al., 1998; Rutherfurd, 2000). Gully severity classes s were defined according to gully size, connectivity to streams, and degree of activity (Whitford et al., 2010). Size referred to the average width of the gully (classified as major when width > 5 m, or minor otherwise), which is linked to the age of the gully and the erosion rate (Table 1). If the gully mouth reached a stream (connected gully), all suspended sediment load eroded from the gully was considered to enter the stream. Conversely, if the gully mouth was removed from the stream network (disconnected), sediments were assumed to be deposited in fans and entrained and transported to the stream by overland flow. Thus, sediment loads eroded in disconnected gullies were multiplied by the hillslope sediment delivery ratio SDR. Inactive gullies are those gullies that are in accretion phase, with vegetation and moss extensively covering the gully floor and walls. Erosion rates of inactive gullies are much lower than actively eroding ones, and were attributed an erosion rate equal to 1/10th of the corresponding active gully class. More details on the gully classes and estimation of erosion rates can be found in Whitford et al. (2010).

Table 1
Extension and erosion rates for major (width > 5 m) and minor (width < 5 m) gully networks in the Avon-Richardson and Avoca catchments.

	Size	Total length (km)	Length disconnected[b] (%)	Length inactive[a] (%)	Current erosion rate[c] (m^2 y^{-1})
Avon-Richardson	Major	170	3	1	0.016
	Minor	197	9	<1	0.021
Avoca	Major	316	80	<1	0.032
	Minor	1687	91	<1	0.051

[a] Inactive gullies present signs of accretion and extensive vegetation cover.
[b] Disconnected – mouth of gullies does not reach the stream network.
[c] Current erosion rate refers to active gullies and is expressed in annual enlargement of cross section.

Gully remediation effects were simulated assuming an effectiveness factor ε of the control works, defined as the ratio of erosion rate with control work over the erosion rate in its absence. The eroding gully length in subcatchment j is thus equal to:

$$L_{s,j} = \Lambda_{s,j} - \varepsilon \lambda_{r,j} \tag{5}$$

where $\Lambda_{s,j}$ is the total length of gullies of severity class s; and $\lambda_{s,j}$ is the length of gully control works.

Streambank suspended sediment load B_j was a function of the river stream power (De Rose et al., 2005; Wilkinson et al., 2009):

$$B_j = (b \rho_w g Q_{BF,j} S_j) \Gamma_j h \rho \Delta \tag{6}$$

where for each subcatchment j, ρ_w is the density of water, g is gravity acceleration, $Q_{BF,j}$ is the bankfull discharge, S_j is the river bed slope, Γ_j is the length of eroding river banks, h is the river bank height, and ρ and Δ are defined as in Eq. (4). The term in brackets represents the bank erosion rate (in m y^{-1}) for the jth subcatchment; the coefficient b is calibrated to give predictions in line with observations (Wilkinson et al., 2009).

Suspended sediment routing and floodplain deposition were defined as in previous versions of CatchMODS, and retained the algorithms of Prosser et al. (2001), reported here for completeness. Total suspended sediment contributions to a reach $X_{\text{in},j}$ were estimated as:

$$X_{\text{in},j} = H_j + G_j + B_j + X_{UP,j} \tag{7}$$

where $X_{UP,j}$ is the suspended sediment received from upstream reaches. Routing to the downstream subcatchment was equal to:

$$X_{\text{out},j} = X_{\text{in},j} - \left(\frac{Q_{OB,j}}{Q_{MAF,j}} X_{\text{in},j} \left(1 - \exp\left(\frac{-\nu A_{f,j}}{Q_{OB,j}} \right) \right) \right) \tag{8}$$

where ν is the settling velocity of sediment particles on the floodplain (m s^{-1}), and A_{fj} is the flood area in the jth subcatchment.

3.5.3. Model coupling–scaling parameters

In coupling the HowLeaky2008 outputs to CatchMODS, upscaling parameters were introduced to account for hillslope processes, i.e. the loss or gain of water and sediments from the field edge to the stream network. In this study, we pursued the simplest possible approach, i.e. using linear ratio scaling factors that were empirically calibrated and validated.

Equation (2) shows that only a fraction of water surplus generated at the point-scale contributes to streamflow. The fraction f is the scaling factor for water. It is noteworthy that runoff and below-root percolation may equally contribute to streamflow; no assumption was made on which pathways (overland flow or interflow) water leaving the field would follow to reach the stream.

Sediment scaling was obtained via the *SDR* factor for hillslope erosion (including disconnected gully erosion) in Eq. (3). Although questionable (e.g. Parsons et al., 2006; Kinnell, 2008), the use of *SDR* as a scaling factor is a common and practical approach for catchment scale sediment load predictions (Lu et al., 2006) and was commensurate with the available data and desired level of complexity of the framework. Although the factor Δ in Eqs. (4) and (6) has a definite physical meaning (i.e. portion of particle size < 63 μm), it lends itself as a scaling factor for gully and streambank erosion, and may potentially correct for uncertainty in the other parameters, particularly erosion rates.

The three scaling factors (*f*, *SDR* and *Δ*) were assumed to be constant in space and independent from discharge and sediment loads. Together with streambank coefficient *b* (Eq. (6)), they were calibrated and validated against observed data at monitoring water quality stations.

3.6. Data collection and organization

Biophysical information required to run the integrated framework can be categorized into climate, topography, soil, land management, and landscape process data.

3.6.1. Climate

Daily time-series for each farming system zone were extracted from SILO data-drill (i.e. spatially interpolated) dataset (Jeffrey et al., 2001) using the centroid coordinates of each farming system zone.

3.6.2. Topography

Topographic information was needed to delineate the subcatchments and to compute subcatchment LS_j factor. Digital Elevation Models derived from 1:25000 maps with pixel size of 20 m were used in both catchments (DPI, 2009). Subcatchment LS_j factor was estimated as the average *LS* factor computed across the subcatchment using the Moore and Wilson (1992) algorithm.

3.6.3. Soils

Published soil data were mapped into groups of similar hydrological behaviour (Melland et al., 2008). The soil column was divided into three layers. Soil parameters were inputs to the HowLeaky2008 model to define water losses by runoff, evapotranspiration, or percolation below the root zone. Most soil parameters were set according to national databases of soil properties (Melland et al., 2008), with the exceptions of CN values, which were set as per Owens et al. (2003), and maximum daily percolation rates (K_{day} mm day^{-1}), which were based on Yee Yet and Silburn (2003) according to soil surface characteristics (hard-setting or not), texture and structure of the soil profile. Table 2 summarizes the soil groups identified in the two catchments and their key soil parameters.

3.6.4. Land use

Publicly available land use maps (DPI, 2009) define agricultural categories too broadly to be useful in assessing soil losses of mixed cropping/livestock farming systems. Farmers and extension officers were asked to map major farming system zones and within each zone to identify current practices and technically feasible alternatives (including crop rotations, tillage operations, pasture types, and grazing management) to reduce sediment loss (Table 3). These alternatives were technically feasible and had been adopted by progressive producers. As such they were considered economically viable but only by producers with sufficient management skills. The information gathered was combined with the land use maps to define current and alternative land managements (Ridley et al., 2008).

3.6.4.1. Avon-Richardson catchment. In the grazing zone of the Avon-Richardson (Fig. 1), current management consists of set stocking on annual plant-based pastures. Perennial pastures occupy about 10% of pasture land. In the mixed cropping zone, annual crops occupy about 40% of the land and livestock grazing (Merino sheep) on annual pastures for the remainder. A standard crop rotation was impossible to define because farmers make decisions about crop sequences on economic grounds, considering agronomic and climatic constraints. In general a four year crop rotation would typically include canola, wheat, barley, and legumes; two tillage operations are usually performed (February and April). In the cropping zone, a 4-year canola-wheat-barley-legume rotation was assumed to occupy most agricultural land, with an exception that due to the recent drought period over the last decade some farmers have returned to the traditional practice of a 2-year rotation of wheat and barley followed by a third year of

Table 2
Hydrologic soil groups of the Avon-Richardson and Avoca catchments, with their extension and key parameters for HowLeaky2008 simulation.

Group name	Dominant soil type (Principal Profile Form; Northcote, 1979)	Extension (% area)	Curve Number (bare soil)	PAWC[a] (mm)	Topsoil percolation rate (mm day^{-1})
Avon-Richardson					
Grey Vertosols	Ug5.24 Ug5.2	11	75	158	24
Hard-setting Grey Vertosols	Ug3.2	10	75	150	12
Heavy Grey Vertosols	Ug5.16	5	75	108	24
Red/Yellow Chromosol	Dr3.41 Dy3.41	7	87	169	30
Red Sodosol (not hard-setting)	Dr5.42	1	85	126	240
Hard-setting Red Sodosol	Dr2.41 Dr2.13 Dr2.23 Dr2.43 Dr3.22 Dy3.42 Ug6.7	41	85	132	30
Crusting Red Vertosols	Ug5.35	21	75	156	24
Shallow Kandosols	Um5.21 Gn2.72 Gn2.81 Gn4.14 Um1.43 Un5.41 Uc4.31	4	75	107	72
Avoca					
Red/Brown Dermosols	Um5.21 Dr2.41	3	65	104	50
Shallow Red/Brown Dermosols	Um5.21 Uc4.31	6	65	39	50
Red Sodosols (on rock)	Dr2.42 Dr2.22 Dr2.41 Dr3.41	21	75	39	50
Red Sodosols (on alluvium or colluvium)	Dr2.42, Dr2.22 Dr2.41 Dr3.41 Dr2.42/3	21	80	105	50
Yellow/Brown Sodosols	Dy3.41 Dy3.42	5	85	96	25
Calcic Red Sodosol	Dr2.23 Dr2.43 Dr3.33 Gn, Uf	33	80	102	25
Calcic Red/Brown Vertosols	Ug6.6 Ug 5.4 Ug5.3	3	85	87	25
Calcarosols (TSL)	Gc1.12	8	65	102	100

[a] Plant Available Water Content.

Table 3
Current and alternative farming systems identified in the two catchments.

Farming system	Current practice	Alternative practice
Avon-Richardson		
Grazing	90% annual pasture; 10% perennial pasture (phalaris dominated)	80% perennial pasture (phalaris dominated), 20% conversion to forest
Mixed cropping	40% broad-acre crop (wheat-barley-fallow rotation); 60% annual pastures	zero tillage, 4-year canola-wheat-barley-legume rotation on cropping land, perennial pastures or lucerne (alfalfa) on grazing land
Cropping	100% broad-acre crop (4-year canola-wheat-barley-legume rotation); occasional wheat-barley-fallow rotation during dry periods	zero tillage 4-year rotation
Avoca		
Cropping	wheat-barley-fallow and opportunistic grazing, 5% grazing (native pastures in the floodplain)	zero tillage and opportunistic grazing
Mixed	40% (50% in the southern part) annual pastures; 60% (50% in the southern part) cropping (wheat-barley-fallow rotation)	perennial pastures (phalaris dominant) on grazing land; zero tillage on cropping land
Grazing (annuals)	20% annual pastures, 80% lifestyle pastures (high ground cover throughout the year due to low stocking rates)	100% perennials (phalaris dominated)
Grazing (perennials)	90% perennial pastures; 10% hay	
Vines and plantations	viticulture and blue gum plantations	

bare soil fallow. The suitable alternative management practices were zero-tillage cultivation for cropping land and perennial grass-based (*Phalaris aquatica*) pastures or lucerne (*Medicago sativa*) for grazing land.

3.6.4.2. Avoca catchment. Several farming system zones were identified (Fig. 3). In the annual pasture grazing areas, an increase in lifestyle farming since the 1990s has improved ground cover throughout the year due to such landholders typically owning few livestock. Mixed cropping systems are prevalent in the majority of the southern and central parts of the catchment. Annual pastures occupy about 50% of land in the mixed southern system and 40% in the central part of the catchment. Continuous cropping in the south is limited to a narrow strip of land adjacent to the Avoca River, but is prevalent in the north. The current crop rotation is typically wheat-barley-fallow, with three tillage operations in the north and no tillage in the south. Vineyards and blue gum plantations are prevalent in the south-western hills. Opportunities for alternative farming systems were identified in perennial pastures for grazing land, and zero tillage with opportunistic grazing in the cropping land.

3.6.5. Landscape processes. Gully extent and type were mapped by extension officers (Whitford et al., 2010). Table 1 summarizes gully network extension and characteristics in the two catchments. In the Avoca catchment, a survey was conducted to map stream banks subject to erosion (Conics, 2009). All maps were digitized and assembled into a Geographic Information Systems (GIS) to input to the catchment model.

3.7. Model calibration and validation

Victoria stream water quality data are published online (DSE, 2010). Daily discharge and approximately fortnightly suspended sediment concentration data from water quality (WQ) monitoring stations in the two catchments were used to estimate annual suspended sediment loads (Table 4; locations of WQ stations are indicated in Figs. 2 and 3). The sample size of suspended sediments was limited and biased toward low flow sampling. Sediment loads were therefore estimated using the ratio method (Letcher et al., 1999), and although the method is well suited for the paucity of the dataset, it may still underestimate loads. In the Avon-Richardson catchment, the sample size of sediment concentration was particularly small (63–93 observations over 14 years, Table 4), therefore data from all the stations were merged to obtain a single ratio value that was applied to discharge data at all five stations. In the Avoca catchment, sample sizes were sufficiently large to derive independent ratios for each station.

All data with the exclusion of the 1980–1989 period were used to calibrate the three scaling factors (f, SDR, and Δ), plus streambank erosion coefficient b in the Avoca; because rainfall amounts and gully erosion rates for the period 1980–1989 represented average conditions of the three decades spanned in the model simulation, 1980–1989 data were chosen to be used for validation.

Model estimates of suspended sediment load depend on the occurrence of overbank floods, which in turn is a function of mean annual flow (Eq. (8)). Model calibration was therefore step-wise: discharge was calibrated first against annual data. Ten-year mean

Table 4
Data availability for water discharge and suspended sediment of the Victorian Water Quality network gauging stations selected for calibration and validation in the two catchments. Numbers in brackets indicate suspended sediment sample size.

Station name	Station ID	Estimated upstream area (km^2)	Data availability	
			Streamflow	Suspended Sediment
Avon-Richardson				
Donald	415257	2275	1990–2006	1994–007 (93)
Banyena	415259	1520	1993–2006	1994–2007 (94)
Wimmera Hwy	415220	589	1971–1999	1993–2007 (63)
Beazleys' Br	415224	259	1969–1987; 1993–1995	n/a
Carrs Plains	415226	121	1963–1999	n/a
Avoca				
Quambatook	408203	3350	1967–2009	1990–001 (135)
Coonooer	408200	2696	1964–2008	1978–2009 (261)
Archdale Junction	408206	658	1987–2008	n/a
Amphitheatre	408202	74	1973–2008	1997–2009 (132)

annual sediment loads (t y^{-1}) were then calibrated for decadal period runs (i.e. one model output for each decade). An evaluation of calibration results was based on linear regression and Nash and Sutcliffe (1970) efficiency of simulations versus WQ station data. Specific sediment yields (t km^{-2} y^{-1}) were also evaluated during the calibration exercise to reduce the bias of calibration toward the downstream stations.

3.8. Sensitivity analysis

Sensitivity analysis identifies parameters that have a large effect on model outputs, and is an important step for model evaluation (Jakeman et al., 2006). If sensitive parameters cannot be accurately identified, model outputs may be highly uncertain and not suited to support decision making.

HowLeaky2008 sensitivity analysis has been reported elsewhere (Littleboy et al., 1999; Melland et al., 2008; Robinson et al., 2010). Runoff and soil loss are most sensitive to ground and canopy vegetation cover as well as several soil parameters affecting the water balance, particularly plant-available water capacity, saturated water content, and CN values. Whilst the coupling of HowLeaky2008 outputs to CatchMODs precluded a detailed sensitivity analysis of catchment-scale outputs to HowLeaky2008 parameters, it is expected that outputs sensitivity to HowLeaky2008 parameters would be smaller at the catchment scale at than the point scale because of the mitigation effect of scaling factors.

Sensitivity analysis of discharge and sediment at the catchment outlets to catchment parameters was limited to a factor screening analysis using the Elementary Effects method (Morris, 1991; Campolongo et al., 2007). Eight k parameters were selected for the analysis (Table 5) in the Avon-Richardson and nine in the Avoca catchment with the inclusion of the streambank erosion coefficient b.

The first three parameters are the scaling factors for water streamflow, hillslope, and gully/streambank erosion. Because of the multiplicative nature of Eqs. (2)–(4), sensitivity for these three parameters can be used as a guideline for any of the other parameters involved in the equations. A similar concept applies to streambank coefficient b. IHACRES parameters (α_q, α_s, and V_s) and the recurrence time for overbank flood t regulate the simulation of bankfull discharge and overbank flooding and volumes. Sediment settling velocity ν is used to estimate the amount of sediment deposition on floodplains (Eq. (8)).

A computational experiment was designed to sample the parameters uniformly within a range of plausible values defined for each catchment. Four p levels of discretization of each parameter were considered. Five r random Morris trajectories were selected following the optimizing sampling strategy proposed by Campolongo et al. (2007), for a total of 45 runs in the Avon-Richardson and 50 runs in the Avoca catchment. The Elementary Effect of the wth input factor is defined as:

$$d_w(X) = \left(\frac{y(X_1, \ldots, X_{w-1}, X_w + E, X_{w+1}, \ldots, X_k) - y(X)}{E} \right) \quad (9)$$

where E was set as $p/2(p-1)$, i.e. 2/3. The sensitivity measure proposed by Campolongo et al. (2007) $\mu*$ is the mean of the absolute Elementary Effects, $|d_w(X)|$ sampled in the computational experiment. A high value of $\mu*$ denotes a high (linear or non linear) sensitivity of the output to the wth factor.

3.9. Qualitative assessment of uncertainty in model predictions

The complexity of any integrated framework associated with limited data availability make identification of key parameters difficult. Although care was taken in the framework to adopt conceptual solutions that minimized the number of catchment parameters, the sheer number of dominant landscape processes versus the limited availability of independent data (i.e. basically only the monitoring stations, Table 4), and the potential for compensating effects among key parameters (e.g. SDR, Δ and coefficient b in Eqs. (3), (4), and (6)) made parameter estimation ambiguous, i.e. several parameter sets may perform equally well (the 'equifinality' problem of Beven, 2000; Beven and Freer, 2001). This is a well-known issue in complex watershed models (e.g. Beven and Binley, 1992; Beven and Freer, 2001; Gallagher and Doherty, 2007; Wagener and Kollat, 2007; Freni et al., 2009). The inability to identify parameters tightly is a source of uncertainty (albeit not the only one, Refsgaard et al., 2007) that propagates to model outputs. Methods that quantify this uncertainty are computationally demanding (Beven and Freer, 2001; Gallagher and Doherty, 2007; Wagener and Kollat, 2007), and prohibitive for application to the framework. Uncertainty of model estimations was therefore addressed qualitatively as follows.

It was recognized that several parameter sets may be behavioural, i.e. may perform similarly in the calibration/validation exercise. Parameter sets were retained as behavioural if they yielded efficiencies in sediment load and specific sediment yield estimation that were not more than 5% lower than the efficiency of the optimal parameter set found during the calibration. The minimum and maximum model realisations obtained using the behavioural parameter sets defined the boundaries of model predictions. As this was not a comprehensive uncertainty analysis, probability was not attached to the result intervals.

4. Results

4.1. Conceptualisation of catchment processes

Conceptual representations of the processes involved in the water and sediment dynamics of the two catchments were captured in schematic diagrams identifying processes by category (hydrology, sediment, and management options; Figs. 2b and 3b). Not all processes were addressed in the framework, for reasons

Table 5
Parameters selected for Elementary Effects sensitivity analysis.

Parameter	Process	Description	Source
f	Hydrology	Fraction of surplus water to streamflow Eq. (2)	Calibration
SDR	Hillslope erosion	Sediment delivery ratio Eq. (3)	Calibration
Δ	Gully/Streambank erosion	Fraction of particle <63 μm Eqs. (4) and (6)	Soil texture analysis; calibration
b	Streambank	Streambank erosion rate (m y^{-1})	Calibration on published erosion rate (De Rose et al., 2005)
α_q	Hydrology	Quick flow decay coefficient	IHACRES calibration
α_s	Hydrology	Slow flow decay coefficient	IHACRES calibration
V_s	Hydrology	Slow flow volume fraction	IHACRES calibration
t	Hydrology	Recurrence time for overbank flood flow (years)	De Rose et al., 2008
ν	Sedimentation	Sediment settling velocity (m s^{-1})	Cheng, 1997

either that discussions with stakeholders indicated them not be sufficiently important, or there was a lack of data or sufficient process understanding to include them meaningfully. Processes not included are on the left side of the major line representing the stream network in the Figs. 2b and 3b. The diagrams were a useful tool for communication with stakeholders, to document the scope of the modelling exercise, and to highlight the management options that were able to be simulated.

4.1.1. Avon-Richardson catchment

Conceptualisation of the Avon-Richardson catchment highlighted that the main stream receives water from outside the catchment through the Wimmera–Mallee Stock and Domestic Supply System just upstream of Banyena WQ station (Fig. 2). Water for domestic use is diverted further downstream outside the catchment. The amount and quality of water supplied and extracted with this system were not available. The hydrology is also further complicated in the lower part of the catchment where surface water can intercept salty water from groundwater aquifers. The groundwater system has been conceptualised (Beverly et al., 2005, 2008), but was not included in this study for reasons of simplicity and a focus on surface driven sediment processes. Because of the introduction and diversion of water through the Domestic Supply System and known surface–groundwater interactions, discharge data from Banyena and Donald WQ stations were not used for streamflow calibration and validation. However, the two stations were included in suspended sediment load calibration, on an assumption that the quality of water introduced and diverted through the Domestic Supply System was of the same quality as the water in the catchment and did not significantly alter the suspended sediment loads transported in the stream network. This assumption was corroborated by the fact that sediment concentrations measured at Wimmera Hwy WQ station were similar to the downstream stations. Hillslope and gully erosion were identified as dominant sources of sediments. Visits and meetings with local experts clarified that streambank erosion was not a dominant process in the catchment; the few eroding reaches were comparable for dimension and hydrology to major gullies and were included in the framework as major gullies.

4.1.2. Avoca catchment

The main hydrologic feature of relevance in the Avoca catchment was the significant loss of water and sediment by overbank flooding in the Lower Avoca River (Fig. 3b). Overbank floods flow unchannelled to several wetlands in the north-west direction. Sediment deposition in these plains is extensive (Rutherfurd and Smith, 1992). Annual discharge at Quambatook is about 40% of discharge at Coonooer, and floodplain losses were adjusted in proportion to the discharge difference at the two stations (GHD, 2008). Gully erosion is extensive in the southern hills, however most gullies are not connected to the streams; sediments are deposited in the flood-out fans at the gully mouths (Rutherfurd and Smith, 1992; Table 1). In contrast to the Avon-Richardson catchment, streambank erosion was considered important relative to other sediment sources and modelled using the streampower approach (Eq. (6)).

4.2. Farming system water and sediment losses (point-scale)

Given the focus on assessing management impacts at the catchment scale, detailed point-scale results for the catchments are not presented. An example of the HowLeaky2008 outputs is shown in Fig. 4. It shows the estimated average annual soil loss for the grazing systems and cropping systems of the mixed cropping area in the Avon-Richardson catchment for the 1950–1999 period. Soil loss (t ha^{-1} y^{-1}) is reported as gross erosion for standard USLE erosion plots (i.e. $LS = 1$, $SDR = 1$ in Eq. (1)).

Perennial pasture systems were estimated to reduce soil loss to levels comparable to forest cover. For cropping systems, due to retention of ground cover, a zero till cultivation substantially reduced soil losses on all soil types. The example illustrates that HowLeaky2008 can quantify soil losses (and water surplus, data not shown) of farm management on different soil types (as well as for climatic conditions, data not shown).

4.3. Catchment model calibration and validation

CatchMODs was run for 10-year periods from the 1960s to the 1990s. Information gathered on the historic land use changes for the different decades (Ridley et al., 2008) was employed to inform the model framework.

4.3.1. Streamflow calibration

Calibration of the water scaling factor f resulted in comparable results for the Avon-Richardson catchment ($f = 0.40$) and the Avoca catchment ($f = 0.45$). Annual simulations were generally good with efficiencies > 0.70 in both the calibration and validation periods (Fig. 5; Table 6). However, streamflow simulations were poorer at Amphiteatre (Avoca), where streamflow was underestimated, and very poor at Carrs Plains (Avon-Richardson), where streamflow was overestimated. Predictions were better for annual than monthly time steps (Table 6), particularly for the Avoca catchment, for which monthly simulations at Quambatook and Archdale Junction (Avoca) were quite poor.

4.3.2. Sediment loads

Calibration of sediment loads (t y^{-1}) resulted in SDR values of 1–2% (Table 7). These SDR values were slightly lower than the hillslope sediment delivery ratios estimated for the two catchments by a study conducted at the Murray Darling Basin scale (Lu et al., 2006), i.e. 2.6% for the Avon-Richardson and 2.4% for the Avoca on average. The differences between calibrated values and Lu et al. (2006) values may indicate that calibration of SDR compensated for errors in estimating hillslope gross erosion, i.e. eq. (1) may overestimate field-scale soil loss in this environment. HowLeaky2008 soil loss estimates were found to be generally in good agreement with long-term net erosion rates estimated with ^{137}Cs technique, although HowLeaky2008 overestimated erosion in one grazing site (Vigiak et al., 2010). Field and catchment scale data were too limited to separate sources of errors in either hillslope gross erosion or delivery to stream. Although lower than expected, calibrated SDR were however not unrealistic for the prevalent flat conditions of the two catchments.

Values for Δ were also comparable among the catchments. In the Avon-Richardson catchment, the calibrated Δ factor of 0.11 was one quarter of the original value (0.44) as estimated from texture analysis of gully wall material (Whitford et al., 2010). Similarly, the calibrated Δ factor for the Avoca catchment (0.15) was lower than the value (0.3) suggested by Rutherfurd and Smith (1992). Calibration of Δ may account for variability in gully wall texture or gully erosion rate. In this respect, it is noteworthy that in the Avon-Richardson an erosion rate equal to a quarter of the value estimated by Whitford et al. (2010) for major gullies still falls within the 95% confidence interval of the erosion rate.

In the Avoca catchment, the calibrated streambank coefficient b (eq. (6)) of 0.001 corresponds to erosion rates of 0.2–2 cm y^{-1} for the eroding reaches, which is lower but comparable to erosion rates reported for similar catchments (Wilkinson et al., 2009), and close to the 1–6 cm y^{-1} estimation of meandering erosion rates based on mean annual flow (Rutherfurd, 2000).

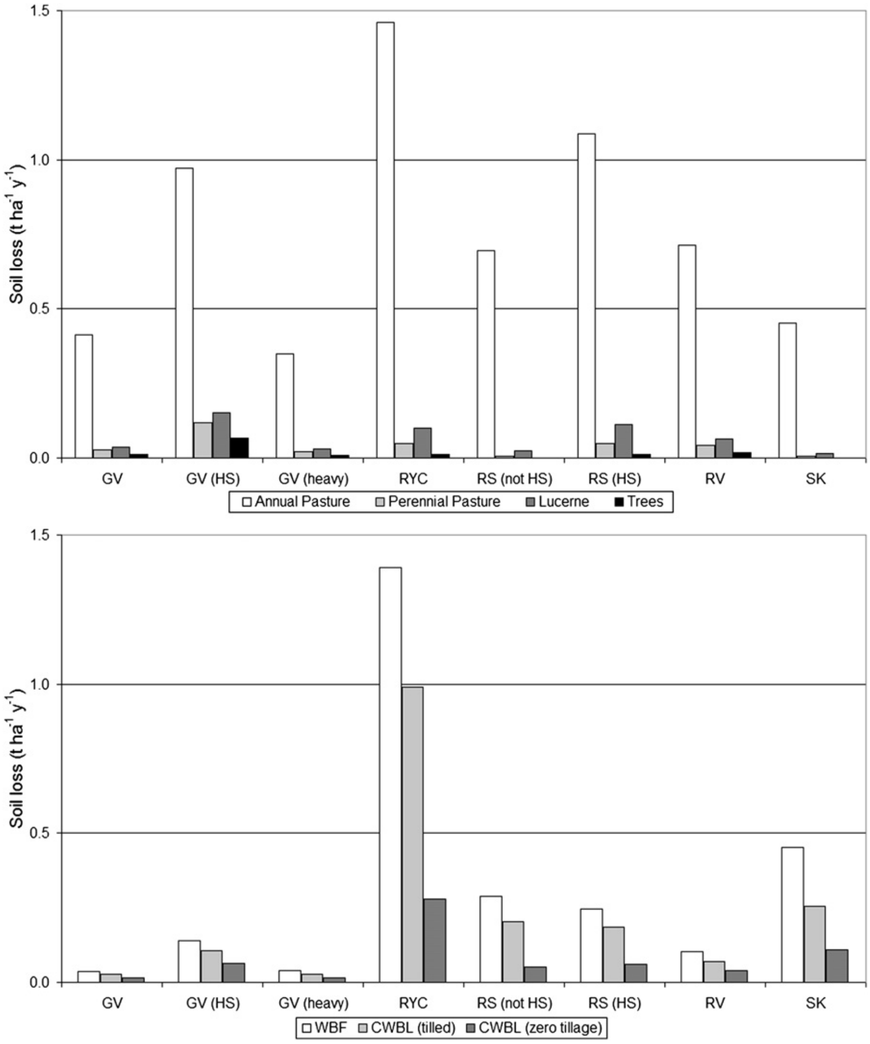

Fig. 4. Impact of land management on average soil loss (t ha^{-1} y^{-1}) on different soils as assessed with HowLeaky2008 for the 1950–1999 period for the mixed cropping areas of the Avon-Richardson catchment: above grazing systems, below cropping systems. GV = Grey Vertosols; GV (HS) = hard-setting Grey Vertosols, GV (heavy) = heavy Grey Vertosols; RYC = Red/Yellow Chromosols; RS (not HS) = not hard-setting Red Sodosols; RS (HS) = hard-setting Red Sodosols; RV = crusting Red Vertosols; SK = Shallow Kandosols. Farm management of cropping systems legend: WBF = traditional wheat-barley-fallow rotation; CWBL (tilled) = current canola-wheat-barley-legume rotation; CWBL (zero till) = zero tillage canola-wheat-barley-legume rotation.

Specific sediment yields may inform of the different sources of sediment, as contributions vary at each station (Wilkinson et al., 2009), and reduce the bias toward downstream stations in calibration. For example, the Amphitheatre station has no streambank erosion contribution, and was used to solve the ambiguity in the calibration of Δ and streambank coefficient *b* in the Avoca catchment. Model simulations of specific sediment yields were close to the 1:1 line in both catchments (Fig. 6), although CatchMODS underestimated specific sediment yields (t km^{-2} y^{-1}) at Carrs Plains (Avon-Richardson) and at Quambatook (Avoca). However, model estimations of specific sediment yields indicate that overall sources of sediments were correctly identified.

4.4. Sensitivity analysis

Not surprisingly, outlet mean annual flow (Q_{MAF}) and mean annual sediment load were most sensitive to catchment scaling factors, as the measure of sensitivity μ* shows (Table 8). In both catchments, mean annual flow was very sensitive to the fraction *f* of water surplus that contributes to streamflow. A change of the water scaling factor *f* by 0.1 changed mean annual flow by more than 30% in both catchments. Conversely, IHACRES flow routing parameters (α_q, α_s, and V_s) were not sensitive in the Avon-Richardson and only slightly sensitive in the Avoca.

Mean annual sediment loads were highly sensitive to the sediment scaling factors *SDR* and Δ. Furthermore, in the Avoca catchment, sediment loads were sensitive to parameters affecting streambank erosion, i.e. streambank erosion coefficient *b*, and to a much lesser extent, water surplus fraction *f* and the return time for overbank flood occurrence *t*.

4.5. Catchment-scale sediment loads and their uncertainty

Model sensitivity to the water scaling factor *f* is of little concern for output uncertainty: monthly and annual discharge data at the monitoring stations are large enough to calibrate and validate the factor *f* with an accuracy of ±0.02 (Fig. 5 and Table 6). Conversely, the limited data available for calibration of sediment loads in comparison with the number of uncertain and sensitive parameters, makes it difficult to identify model parameters tightly. In both catchments, several parameter sets could match sediment loads and specific sediment yields observed at the WQ gauging stations, i.e. were behavioural. The issue was particularly acute in Avoca catchment where three possible sources of sediments

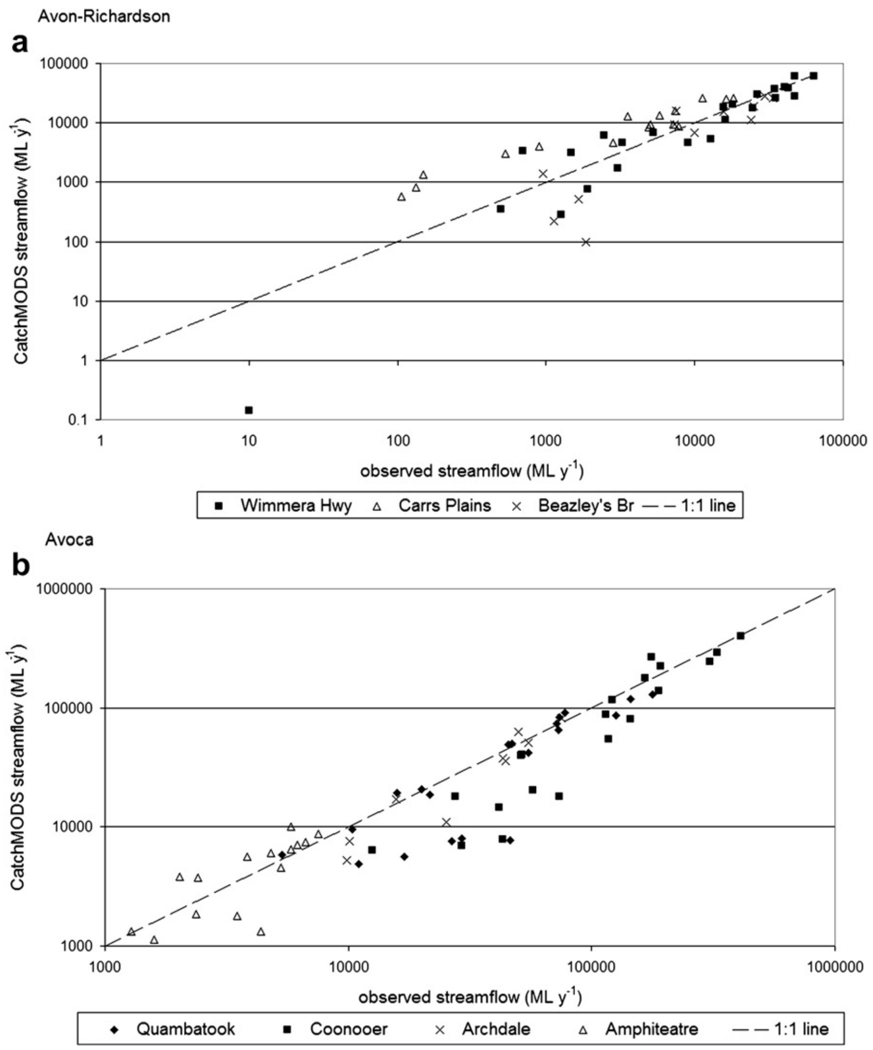

Fig. 5. Calibration of annual water discharge in the two catchments: a) Avon-Richardson, b) Avoca.

(hillslope, gully, streambank) had to be calibrated against three monitoring station data (nine data entries). An additional complication is that uncertainty in the estimates of suspended sediment loads at the gauging stations is high and may likely exceed 20% of load values (Rode and Suhr, 2007; Harmel et al., 2009). In the Avon-Richardson catchment, behavioural parameter sets lied within the boundaries of [$SDR = 1.8\%$; $\Delta = 0.13$] and [$SDR = 2.7\%$ and $\Delta = 0.07$]. In the Avoca, behavioural parameter

Table 6
Calibration and validation results (linear regression equation and Nash-Sutcliffe efficiency) of observed vs predicted annual (A) and monthly (M) streamflow at the selected WQ stations of the study areas. n = sample size; R^2 = adjusted regression coefficient; E = Nash-Sutcliffe efficiency.

WQ station		Calibration				Validation (1980–1989)			
		n	Equation[a]	R^2	E	n	Equation	R^2	E
Avon-Richardson									
Wimmera Hwy	A	27	$y = 0.94x$	0.89	0.90	10	$y = 0.79x$	0.26	0.69
	M	313	$y = 0.77x$	0.77	0.78	120	$y = 1.10x$	0.70	0.74
Beazley's Br	A	14	$y = 0.80x$	0.84	0.82	10	$y = 0.74x$	0.65	0.74
	M	162	$y = 0.63x$	0.75	0.73	90	$y = 1.26x$	0.74	0.74
Carrs Plains	A	19	$y = 1.65x$	0.86	−0.10	10	$y = 1.39x$	0.35	−0.12
	M	225	$y = 1.21x$	0.63	0.26	120	$y = 0.59x$	0.69	0.29
Avoca									
Quambatook	A	20	$y = 0.81x$	0.88	0.76	10	$y = 1.09x$	0.91	0.89
	M	240	$y = 0.88x$	0.48	0.33	120	$y = 0.72x$	0.55	0.58
Cooner	A	20	$y = 1.04x$	0.86	0.87	10	$y = 0.97x$	0.86	0.90
	M	240	$y = 0.79x$	0.65	0.69	120	$y = 0.79x$	0.72	0.75
Archdale	A	10	$y = 1.04x$	0.74	0.77	n/a			
	M	120	$y = 0.87x$	0.54	0.47	n/a			
Amphitheatre	A	16	$y = 0.80x$	0.60	0.64	10	$y = 0.70x$	0.90	0.61
	M	183	$y = 0.64x$	0.64	0.52	120	$y = 0.57x$	0.58	0.57

[a] y = CatchMODS streamflow, x = observed streamflow.

Table 7
Results of sediment load calibration and validation in two Victorian catchments. *SDR* = sediment delivery ratio; Δ = gully/streambank erosion fraction contributing to suspended sediment. R^2 = adjusted regression coefficient; E = Nash-Sutcliffe efficiency.

		Avon-Richardson	Avoca
SDR (%)		2	1.25
Δ		0.11	0.15
Streambank coefficient *b*		n/a	0.001
Calibration	n	11	6
	Equation[a]	$y = 0.98x$	$y = 0.91x$
	R^2	0.80	0.84
	E	0.74	0.82
Validation	n	3	3
	Equation[a]	$y = 0.82x$	$y = 0.96x$
	R^2	0.96	0.95
	E	0.91	0.95

[a] y = CatchMODS sediment load (10-year annual average, t y^{-1}), x = observed sediment load.

sets comprised *SDR* in the range 0.5–1.5%; Δ in the range 0.10–0.24, and coefficient *b* in the range 0.0001–0.0025 (equivalent to mean streambank erosion rates from nil up to 5 cm y^{-1}). These envelopes of parameter boundaries were used to assess the model output boundaries, as reported in brackets.

Suspended sediment load at the outlet of the Avon-Richardson catchment for the 1990–1999 decade was estimated to be 3350 (3300–3700) t y^{-1} (equal to 1.2 t km^{-2} y^{-1}). Hillslope erosion was up to 3.7 t km^{-2} y^{-1}, and contributed 65% (60–80%) of suspended sediment loads at the terminal Lake Buloke. Gully erosion specific sediment yields were up to 3.7 t km^{-2} y^{-1}, and contributed 35% (20–40%) of suspended sediments reaching Lake Buloke. Contributions of the two sources varied between subcatchments (Fig. 7), however both hillslope and gully erosion were highest in the southern grazed hills.

Suspended sediment loads for the Avoca catchment were estimated to be 4000 (3500–5100) t y^{-1} (equal to 1 t km^{-2} y^{-1}). This lower load than for Avon-Richardson accounts for the strong sedimentation due to overbank flood depositions in the lower Avoca catchment. Suspended sediment source contributions changed with parameter sets in the range 5–24% of hillslope erosion, 55–93% of gully erosion, and 1–21% of streambank erosion. Hillslope specific sediment yields were up to 2.3 t km^{-2} y^{-1}, being highest in the western central part of the catchment, with slopes >30% and low vegetation cover under annual pastures. The 90th percentile of gully-specific-sediment yield was 5 t km^{-2} y^{-1} (up to 40 t km^{-2} y^{-1}); gully erosion was highest in the major gully systems in the central-western part of the catchment. Streambank-specific-sediment yields were up to 3.8 t km^{-2} y^{-1} where the southern headwater creeks were very active (Fig. 8).

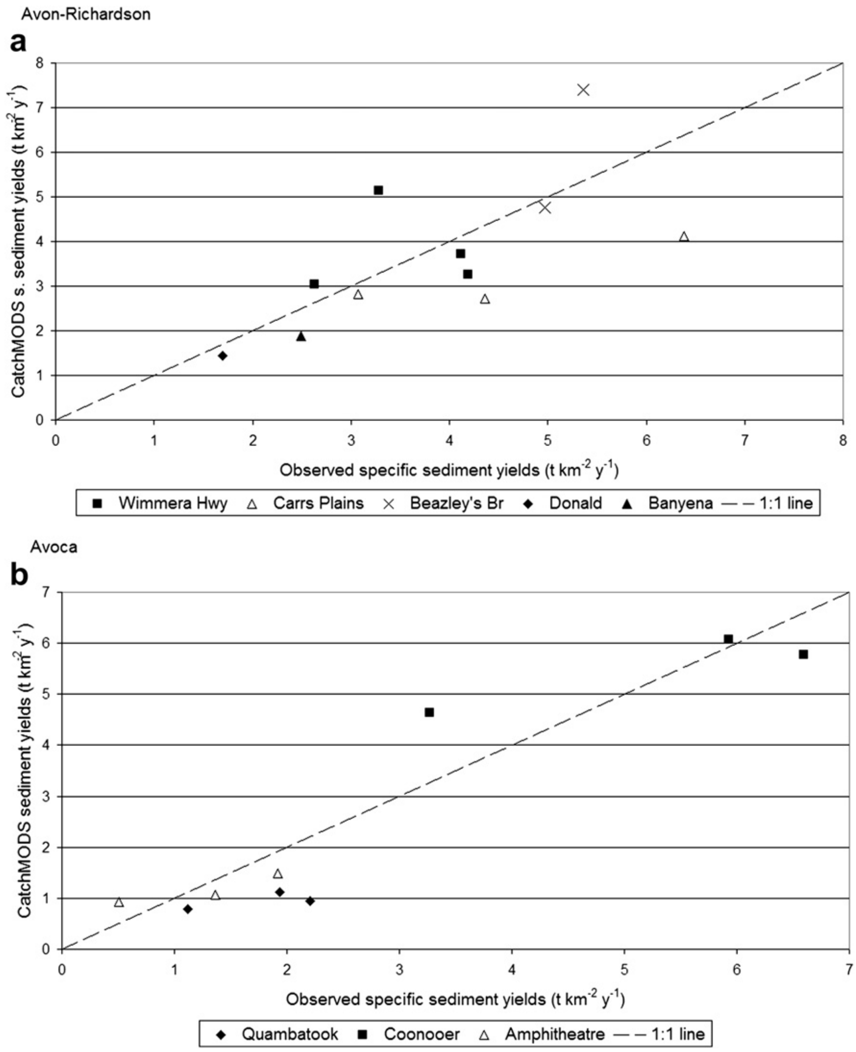

Fig. 6. Predicted and observed specific sediment yields (t km^2 y^{-1}) for each decade of model simulation per WQ station: a) Avon-Richardson, b) Avoca.

Table 8
Sensitivity of mean annual discharge (Q_{MAF}) and mean annual sediment load at the catchment outlets to catchment-scale parameters. Sensitivity is expressed as the average absolute elementary effect μ* (Campolongo et al., 2007) per step of change in each parameter.

		Tested range	Step of change	Avon-Richardson		Avoca	
				Q_{MAF}	Sediment	Q_{MAF}	Sediment
Unit of measure				ML y^{-1}	t y^{-1}	ML y^{-1}	t y^{-1}
Base results (from calibration as in Table 7)				55300	3350	56000	4000
f	fraction	0.3; 0.55	0.1	20218	0	18144	301
SDR	fraction	0.005; 0.05	0.01	0	1653	0	1093
Δ	fraction	0.05; 0.35	0.1	0	1573	0	4864
b		0.00001; 0.005	0.001	n/a	n/a	0	1300
α_q		−0.05; −0.60	0.01	0	0	4	4
α_s		−0.90; −0.99	0.01	0	0	4	1
V_s	fraction	0.0; 0.2	0.1	0	0	12[a]	4[a]
t	year	1.5; 3.0	1	0	0	0	118[a]
ν	m s^{-1}	10^{-6}; 10^{-3}	10^{-4}	0	0	0	0

[a] Denotes negative sensitivity to the parameter.

4.6. *Impact of management strategies on catchment sediment reduction*

The modelling framework can be used to assess the impact of management for any subcatchment. For the purpose of illustration, management impacts are presented on a catchment-wide basis for the period 1990–1999 (Fig. 9).

In the Avon-Richardson catchment, three management scenarios were modelled. An alternative farming system management scenario (ALT) consisted of the alternative farm management practices shown in Table 3. Gully remediation through fencing and revegetation could focus on either major (G1) or minor (G2) gullies. The most effective means for reducing the sediment load at the outlet was the alternative farming system (ALT), which was simulated to achieve a reduction of about 1700 t y^{-1} (50% of the total current suspended sediment load). In contrast, if management actions were focussed on gully remediation, protecting minor gullies appears to be more effective than focusing on major ones, achieving a reduction of 450 t y^{-1} (14%). Remediation of all gullies would yield the combined effect of G1 and G2, with a total reduction of 800 t y^{-1} (24%).

In the Avoca catchment, management scenarios were defined as: alternative farming practices (ALT) as defined in Table 3; fencing and revegetation of connected gullies (G1) or disconnected gullies (G2), and fencing and revegetation of stream banks (STR). The alternative farming system scenario (ALT) could reduce suspended sediment loads by 550 t y^{-1} (14% of current load); similar results were suggested through remediation of streambanks (STR). Notwithstanding uncertainty considerations, remediation of connected gullies (G1) yielded the largest impact on suspended sediment load reduction, estimated at 1770 t y^{-1} (44% of current load) whereas remediation of disconnected gullies (G2) had a negligible impact on sediment reduction. This result concurs with Rutherfurd and Smith (1992) who also concluded that disconnected gullies did not contribute much sediment to streams.

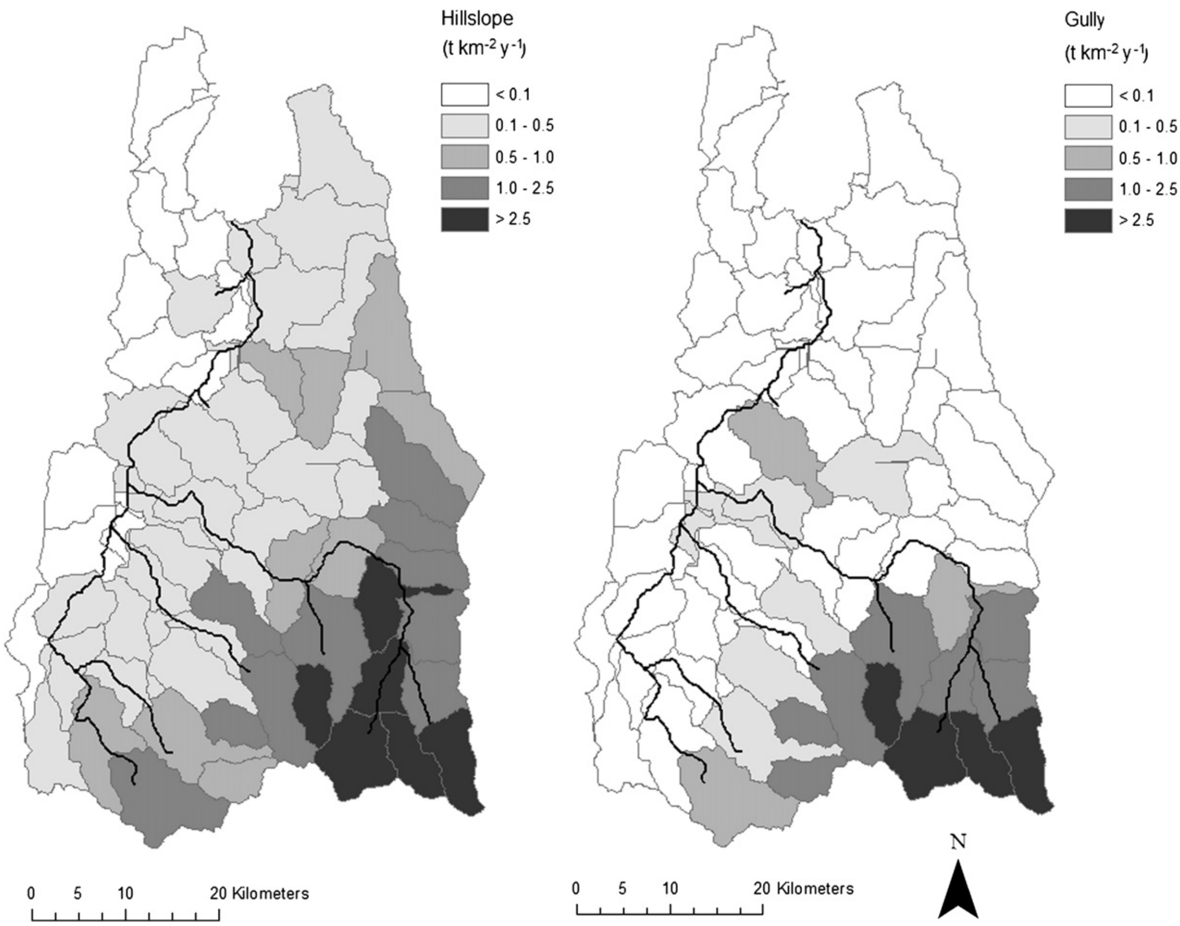

Fig. 7. Estimated specific sediment yields (t km^2 y^{-1}) from hillslope and gully erosion in the Avon-Richardson catchment in the 1990–1999 decade.

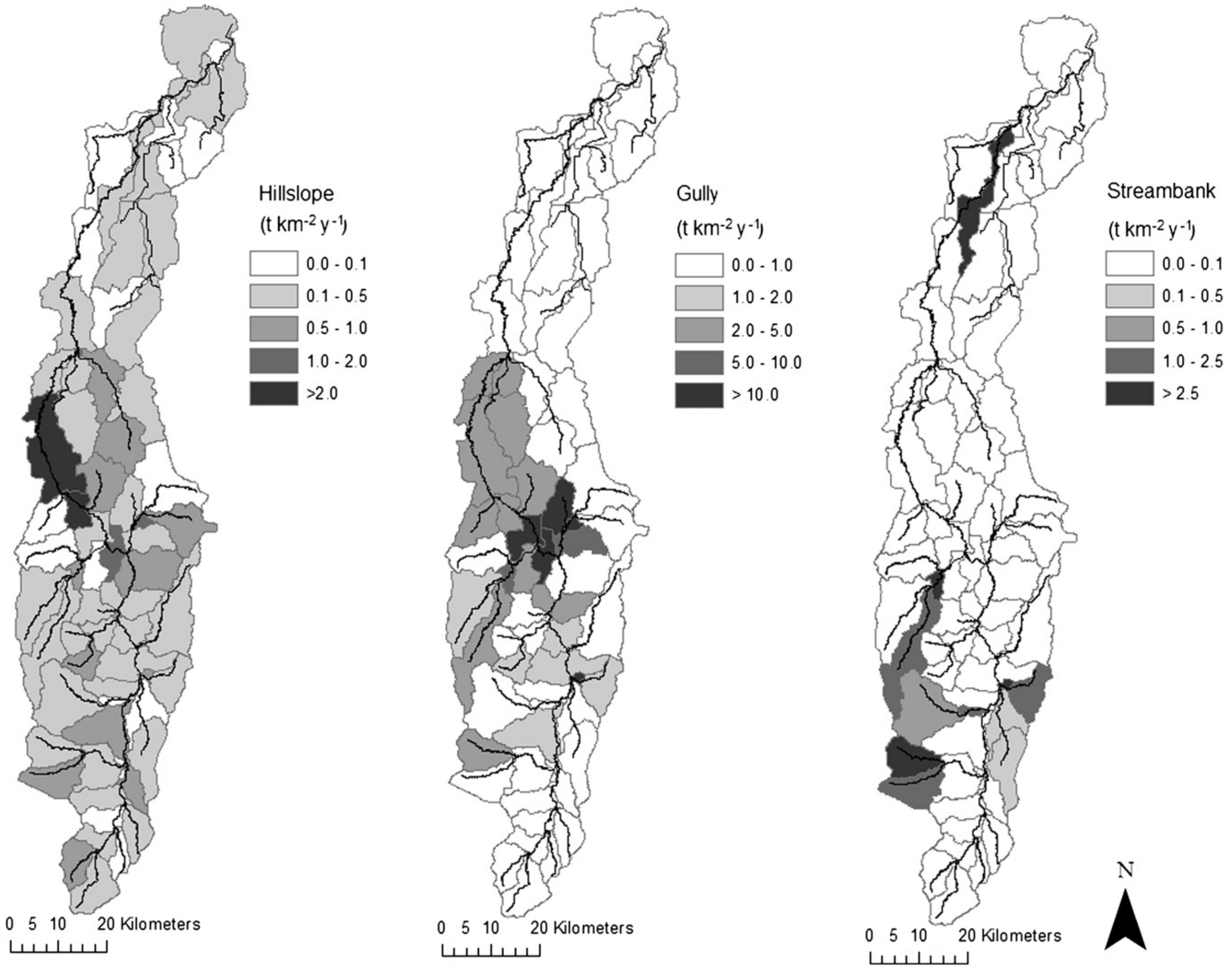

Fig. 8. Estimated specific sediment yields (t km^2 y^{-1}) from hillslope, gully, and streambank erosion in the Avoca catchment for the 1990–1999 decade.

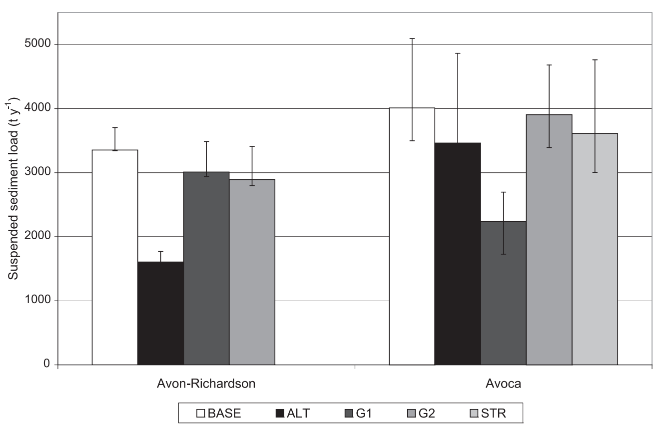

Fig. 9. Estimated impacts of management scenarios on suspended sediment load at the catchment outlets. ALT = alternative faming system; $G1$ = 1st gully scenario (i.e. remediation of major gullies in the Avon-Richardson or connected gullies in the Avoca); $G2$ = 2nd gully scenario (i.e. remediation of minor gullies in the Avon-Richardson or disconnected gullies in the Avoca); STR = remediation of stream banks. Error bars indicate minimum and maximum suspended sediment load estimated in the range of behavioural parameter sets.

5. Discussion

5.1. Suitability of the framework to assess management impact on suspended sediment load

Decision support systems can help decision-makers evaluate the likely impact of targeted environmental investments and policies. Although integrated models are simplified representations of complex systems, and model outputs result from complex interactions regulated by largely uncertain parameters, they provide a transparent framework to account for the dominant sources of contaminants in a given system. The work has facilitated a more evidence-based and constructive debate by staff and landholders within the North Central Catchment Management Authority about investment strategies to improve environmental outcomes.

Environmental Best Management Practices (BMPs) are often claimed to be the major solution to diffuse source environmental pollution. The reality is that the impact of farm management choices will depend on the environmental conditions; for example the presence of depositional areas may reduce the sediment delivery to stream, thus the impact of farm management on downstream reaches may be negligible. The use of point-scale models such as HowLeaky2008 may help explore where farm management systems can reduce environmental pollution, and, at least in relative terms, by how much (e.g. Freebairn et al., 2003;

Rattray et al., 2004; Robinson et al., 2010; White et al., 2010). Fig. 4 shows that even in the same climatic conditions, soil losses may be very different depending on soil characteristics.

Gully and streambank erosion can be dominant sources of sediments (e.g. Hancock et al., 2007; Wilkinson et al., 2009). CatchMODS was developed to assess land use impact at the catchment scale, but previous versions had little real capability to explore effectiveness of alternative farm management systems. The main improvement of the HowLeaky2008-CatchMODS framework lies in the ability to assess the impact of farm management in different environments, and sets these impacts into the context of the catchment landscape processes.

The data requirements for the framework are onerous, yet achievable beyond a research project domain, e.g. by a catchment management agency. Spatial information on climate, soil, land use and topography is commonly available in Australia, although soil information in particular is rarely in a form immediately suitable as model inputs. Careful interpretation of data for the model application is essential, particularly for soil hydrology (Melland et al., 2008). Other environmental data, such as distribution of eroding gullies or streambanks and their erosion rates are usually more rarely quantified or available, although Whitford et al. (2010) have recently developed a method for rapid assessment of gully erosion. Data on streambank erosion are also often lacking (Rutherfurd, 2000), however equations to assess streambank erosion rates based on discharge have been developed for southeast Australia (Rutherfurd, 2000; De Rose et al., 2005). Further calibration of such equations would be useful in a range of catchments. Mapping of erosive river reaches can be done either using remote sensing (e.g. Hancock et al., 2007; Wilkinson et al., 2009) or field surveys (e.g. Conics, 2009). Despite greater data requirements than currently being used by public agencies to make investment decisions, additional data collection is not overly onerous and is likely to result in much greater confidence about the sources and relative contributions of sediment sources.

5.2. System quality

An essential attribute of a good decision support system is the system quality (Volk et al., 2009). The quality of the framework components has been reported previously (see Melland et al., 2010; Vigiak et al., 2010 for HowLeaky2008; Whitford et al., 2010 for gully erosion; De Rose et al., 2005; Wilkinson et al., 2009 for streambank erosion; Newham et al., 2004; Prosser et al., 2001 for deposition and sediment routing). The model kernel is considered appropriate for the spatial and temporal resolution sought in the integrated framework.

Coupling the models at two scales required several assumptions affecting the overall system functioning. In this work, the simplest possible up-scaling approach was adopted, i.e. by assuming spatially-constant, linear scaling factors. This approach is common (Hansen and Jones, 2000), and mainly dictated by the limited data availability to support more sophisticated approaches. The downside of adopting such a simple approach is that the hillslope hydrology impacts on water and sediment delivered to the stream were not fully considered. More sophisticated approaches to implement spatially-variable scaling factors inspired by regionalisation approaches have been trialled elsewhere (Vigiak et al., 2008; 2009) but the increased detail in representing hillslope processes came at the cost of a much higher number of spatially-variable parameters. If there is supporting data, spatially-variable approaches to refine hillslope process representation could be easily implemented in the framework. In the two study areas water quality data were too limited to warrant this additional complexity. The risk of over-parameterisation was already present even using spatially-constant factors, and ambiguity existed in the identification of *SDR*, Δ and coefficient *b* factors from the limited sample of sediment loads at WQ stations.

The model was very sensitive to key catchment parameters such as *SDR*, Δ and streambank erosion coefficient *b*. This feature, however, is not unique to the integrated framework developed in this study, being common in other catchment models of similar structure (Newham et al., 2003; Hancock et al., 2007; Wilkinson et al., 2009). The point-to-catchment model coupling introduced only the water scaling factor *f* as a new parameter, while the *SDR* parameter for point soil loss scaling is common to many catchment erosion models. Annual discharge data are usually sufficient to identify the factor *f* tightly; however the assumption is that parameterisation at the point scale (HowLeaky2008) will capture the differences in hydrological behaviour between soil types, at least in relative terms. Poor model predictions at Carrs Plains (Avon-Richardson; Table 6, Figs. 5 and 6) may indicate that this might not have been the case for the soil types upstream of this station.

Identification of sensitive parameters relies on calibration against observed discharge and sediment loads. Availability of monitoring data is therefore crucial for enhancing the reliability of model outputs. Public availability of water quality data in Victoria is commendable, and indeed almost a luxury, however monitoring of water quality is still much too small and scattered, hindering a thorough evaluation of distributed catchment–scale models. The Avoca catchment results are particularly illustrative, where only one water quality monitoring station had more than 12 years of sediment concentration monitoring. This data paucity represents the general situation under which public agencies are operating to make environmental management investment decisions. Given the water quality problems in many Australian rivers, modelling would benefit from improvements of long-term water quality monitoring programs. Furthermore, where available, independent data such as sediment tracing (e.g. Hancock et al., 2007; Fu et al., 2009), sediment-hydrograph analysis (Picouet et al., 2001), or local field investigation may improve identification of parameters and sources of sediment, and thus reduce model prediction uncertainty.

Sediment load estimations reported in this study differ markedly from previous nation-wide estimates for the study areas (Australian Government, 2008), both in terms of specific sediment yields and partitioning among sediment sources. Given that the national assessment was conducted using broad scale land use maps, without locally available data and without local calibration, our estimates are likely to be locally more realistic and accurate than the previous work.

The present study highlighted two major sources of uncertainty that may affect management scenario analysis, i.e. data paucity for calibration and validation, and ambiguity in identification of key parameters. The impact of these combined uncertainties on framework outputs affecting management decision-making was found to be limited, i.e. neither location of high priority areas (Figs. 7 and 8), nor the ranking of most effective management scenarios (Fig. 9) changed within the uncertainty boundaries that were identified. However, several other sources of uncertainty that were not explicitly addressed in this study may affect management decision-making. Some of these uncertainties lie within the assumptions taken in the framework development, such as efficiency of gully or streambank remediation work, or uncertainties in input data. These sources of uncertainty could be addressed by setting appropriate monitoring programs or surveys by catchment management agencies. Albeit important, these sources of uncertainty may be minor compared to sources of uncertainties that are instead external to the framework, for example issues of ecological nature, e.g. temporal lags between adoption and environmental outcomes or ecological relationships between sediment loads and pollutant concentrations (e.g. Lynam et al., 2010); or socio-economic uncertainties, such as likely adoption rates of environmental BMPs under voluntary or incentive-driven schemes (Cary and Roberts, 2009). Socio-economic and ecologic uncertainties will greatly affect final management strategies. The HowLeaky2008-CatchMODS framework does not address all key issues that decision-makers need to account for in designing effective natural resource management strategies, however it enables assessing quantitatively the impact of farm management at the catchment scale and locating priority areas for intervention, and therefore represents an improved tool to assist natural resource managers.

5.3. Other features of the framework

Conceptualisation of the system is essential to build confidence in model output analysis and to draw the boundaries of the modelling scope. It is an important phase to engage with stakeholders and build a shared understanding of the system (Jakeman et al., 2006; Croke et al., 2007; Lynam et al., 2010). The modular coding platform of CatchMODS allowed maintenance of transparent relationships between the system concepts and model code. It also ensured sufficient flexibility to allow for including a level of complexity commensurate to process understanding and data availability (see Argent et al., 2009, for an extensive discussion on a similar modular platform). The framework's flexibility was well tested in this study, being initially developed for the Avon-Richardson catchment, and then readily adapted to the Avoca catchment. Flexibility in the scenario analysis is also illustrated through application in the two catchments, where priorities for management could focus on different gully classes, and/or hillslope or streambank erosion (Fig. 9), or spatially (Figs. 7 and 8).

Whilst this study did not evaluate the perceived usefulness or the user satisfaction of the framework (Volk et al., 2009), both models were created to engage with stakeholders throughout the modelling development. HowLeaky2008 and CatchMODS were already equipped with user-friendly interfaces that allowed visualisation of results (Freebairn et al., 2003; Newham et al., 2006) which were retained in the integrated framework and facilitated stakeholder engagement.

5.4. Major limitations

There are some important limitations of the integrated framework. Whilst spatial aggregation of HowLeaky2008 HRUs into CatchMODS subcatchment was effective to assess farm management scenarios, it precluded consideration of spatial targeting within a subcatchment aimed at particular land management and soil type combinations, which could improve the effectiveness of farm management actions relative to gully or streambank sources. Whilst the impacts of land management–soil combination can be assessed with HowLeaky2008 (Fig. 4), implementation at the catchment scale smoothes the effect of land management among all soil types within a subcatchment. In addition, the one-way, offline coupling precluded the propagation of point-scale parameter uncertainties to the catchment scale outputs, hindering a comprehensive evaluation of model uncertainties.

The framework outputs can be visualised in maps (Figs. 7 and 8), however the current CatchMODS platform does not have (nor require) a GIS interface. Whilst this is an advantage for model end-users who may not have GIS data or capability, a major disadvantage is that the preparation of catchment inputs is cumbersome and time-consuming.

5.5. Future development needs

Suspended sediments are one of several important aspects of stream water quality; nutrient concentrations and loads are also of major concern for aquatic ecology and river health (Cerco et al., 2010; Lynam et al., 2010; Mayorga et al., 2010). Because suspended sediments are nutrient vectors, an integrated framework that simulates sediment generation and transport is an important step towards inclusion of additional complexity to also account for nutrients. Both HowLeaky2008 and CatchMODS are equipped with algorithms for generation and transport of dissolved and particulate phosphorus (Robinson et al., 2007; Newham et al., 2004) and future research will include assessment of the ability to model phosphorus transport at the field and catchment scale.

Analysis of 'what-if?' scenarios such as those tested in the two catchments can facilitate a more evidence-based and constructive debate on alternative strategies for environmental outcomes. In assessing the most cost-effective management strategies, it would be useful to better consider the costs and benefits of different strategies. CatchMODS is equipped with a simple module to compute costing of the intervention strategies for gullies and streambanks (Newham et al., 2004; Volk et al., 2009). This is currently deliberately simple, however, the modular coding platform could be adapted to host more complex economic evaluations of management scenarios. This would be useful to explore to assist public agencies make more cost-effective management decisions across gully, streambank and farm management options.

6. Conclusions

Application of the framework to two catchments in Victoria, south-eastern Australia showed the approach was suitable to assess the impacts of farm management choices on soil loss in different environments, and put them in the catchment context, therefore accounting also for dominant landscape erosion processes, be they hillslope, gully, and/or streambank erosion.

The framework is more suitable than either point or catchment models used singly to facilitate discussion on how to improve water quality at the catchment scale. Several features make it valuable for use by public investment agencies, such as Australian catchment management agencies. These include that: (i) farming system options and landscape processes are integrated into a conceptual system that allows comparison of management alternatives; (ii) the framework maintains a high level of transparency and is equipped with a user-friendly interface to facilitate stakeholder engagement; (iii) the modular structure of the code platform makes it flexible and able to be adapted for other regions.

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