Estimating soil erodibility for the RUSLE with rainfall simulation in central Queensland, Australia

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ABSTRACT

Context. Land use change in central Queensland has led to increased sediment and nutrient loads in the Great Barrier Reef (GBR) lagoon, impacting water quality. Accurate soil loss prediction is essential for managing these impacts, necessitating improved erodibility estimates for grazing lands. **Aims.** The objective of this study was to advance the prediction of fine sediment exports to the GBR lagoon from hillslope sources. It specifically aims to enhance hillslope erosion modelling by refining the Revised Universal Soil Loss Equation (RUSLE) soil erodibility estimates (K-factors) for grazed lands. **Method.** An integrative approach was employed, combining the empirical RUSLE with the process-based Water Erosion Prediction Project (WEPP). This study leverages observed soil loss data from simulated rainfall experiments to calibrate WEPP and produce an annual average soil loss to substitute into RUSLE. Prevalent soil types in the Fitzroy Basin were studied. **Key results.** The WEPP was calibrated with observations and shown to be able to simulate observed sediment loss with good performance indicator values (R^2 = 0.9, PBIAS = 1.9%, and NSE = 0.87). The study identified that the existing RUSLE nomograph-derived K-factors would overestimate soil erodibility by up to 63% for the soils evaluated. Traditional methods predict higher erodibility; however, results from this study classify these soils as having low to moderate erodibility, attributed to local soil consolidation and limited erosion detachment processes typical in grazed areas. **Conclusions**. Findings suggest that the conventional RUSLE erodibility nomograph does not adequately reflect the erodibility of consolidated or highly aggregating grazing soils found in the GBR catchments, leading to overestimated K-factor values and, subsequently, overestimated contribution of hillslope erosion to the sediment budget for the GBR catchments. **Implications.** This research contributes to delivering a cost-effective, measurement-based method for K-factor estimation to improve hillslope erosion prediction for grazing lands in central Queensland.

Keywords: erosion, grazing, Great Barrier Reef, hillslope erosion, rainfall simulation, RUSLE, soil erodibility, WEPP.

Introduction

Anthropologic land use changes in the Great Barrier Reef (GBR) catchment of Queensland, Australia, have resulted in declining water quality due to increased sediment loads caused by reduced vegetative surface cover, application of fertilisers and pesticides, and agricultural tillage and grazing pressures (Seabrook *et al.* 2006; Packett *et al.* 2009; Carroll *et al.* 2012; Brodie *et al.* 2013; Lewis *et al.* 2021; Waterhouse *et al.* 2024). Management of these sediment loads is critical to the health of the GBR, with sediment load targets for each of the 35 GBR catchments outlined in the Reef 2050 Water Quality Improvement Plan (WQIP) (The State of Queensland 2018). To understand the relationship between upland erosion processes and fine sediment discharge at the end of catchment, modelling frameworks are used to predict sediment exports against monitored loads. Estimates of fine sediment loads discharged to the GBR lagoon, and their likely sources and dominant erosion processes contributing to end-of-catchment loads, are of particular importance to management of the GBR as a natural resource. Sources of sediment relevant to the GBR are defined as surface (hillslope) or subsurface (gully and streambank).

The contribution of sediment from grazed land in the GBR catchment increased in the late twentieth century (Bainbridge *et al.* 2018; Bartley *et al.* 2018; Wilkinson *et al.* 2018; Lewis *et al.* 2021; McCloskey *et al.* 2021); modelling has suggested that grazing accounts for 60% of the fine sediment exported to the GBR lagoon from surface and subsurface sources (Prosser and Wilkinson 2024). Modelled fine sediment exported from the 156,000 km^2 Fitzroy catchment within the GBR catchment area (423,000 km^2) is over 1300 kt year^{-1}, where 73% of the land use in that catchment is grazed (Prosser and Wilkinson 2024).

Soil losses from grazing land in the GBR catchment are modelled with the GBR Dynamic SedNet catchment model (Dynamic SedNet). This model facilitates understanding of major sediment sources, dominant erosion processes, and associated land-use driving these processes (McCloskey *et al.* 2021). Conceptually within Dynamic SedNet, the sources of sediment are classified as surface and sub-surface to represent either the disturbance of the upper surface profile or the deeper soil profiles, respectively (McCloskey *et al.* 2021). Recent Dynamic SedNet estimates indicate that hillslope erosion is exporting 22% of the total sediment from the GBR catchment (McCloskey *et al.* 2021; Prosser and Wilkinson 2024). Predicting erosion from non-point and often diffuse hillslope sources in the GBR grazing land within Dynamic SedNet is based on the empirical Revised Universal Soil Loss Equation (RUSLE) (Waters *et al.* 2014; Ellis 2018; McCloskey *et al.* 2021).

The RUSLE is a simple, convenient, erosion prediction model that includes a range of factors and conditions, most of which can be confidently evaluated and reliably applied (Wischmeier 1976; Renard *et al.* 1997). The RUSLE has five major empirical factors used to predict soil loss: rainfall erosivity, soil erodibility, slope length and steepness, cover, and practice. Soil erodibility (K), the susceptibility of a soil to erosive forces, is an important soil-related factor considered in the RUSLE and is defined as the rate of soil loss per unit of rainfall erosivity for a bare unit plot (Renard *et al.* 1997). Erodibility in the RUSLE equation is rarely field-measured and is estimated using an empirical nomograph equation based on soil properties including organic matter, soil texture, structure, and permeability properties (Wischmeier and Mannering 1969; Loch and Rosewell 1992; Silburn 2011). In Dynamic SedNet, soil erodibility is calculated using soil properties in a state-wide soil database using a modified nomograph (Loch *et al.* 1998; Lu *et al.* 2003; McCloskey *et al.* 2021).

The nomograph method for estimating RUSLE erodibility was originally developed on cultivated croplands and construction sites in the USA on medium-textured (silty) soils by Wischmeier and Smith (1978), and is a widely used alternative to direct measurement of soil erodibility. This was often made necessary due to the resources and time required to measure the mean annual soil loss at the plot scale over a period of at least 20 years (Renard *et al.* 1997). Measuring sediment

loss from a catchment or USLE plot is both time-consuming and costly, and therefore rarely done here in Australia. The nomograph method for estimating soil erodibility from measurable soil properties has been found to perform poorly on Australian soils with a high clay content (Loch and Rosewell 1992), nor has the nomograph been assessed in Australia for soils under grazing (Loch 2015a). Loch and Rosewell (1992) also raised concerns with the clay fraction represented in the original nomograph, suggesting that this size class is more likely present within the soil as aggregates within the range defined for silt and very fine sand. Other factors for resistance, or susceptibility of a particular soil to erosion, include pre-existing conditions such as organic carbon content, soil strength, texture, moisture conditions, and aggregate stability (Wischmeier and Mannering 1969; Young and Mutchler 1977; Coote *et al.* 1988; Kok and McCool 1990). The erodibility nomograph is therefore likely to be applicable only to soils similar to those considered in the original USA dataset (Loch and Rosewell 1992; Ciesiolka *et al.* 1995).

Loch and Rosewell (1992) also made the point that the nomograph equation does not consider the impact of consolidation of the soil surface due to surface sealing under repeated rainfall events. This finding led to the development of a modified nomograph equation, K_m. Research by Brooks *et al.* (2014) measured hillslope erosion across grazed savannah in the Wet Tropics (Cape York catchment) and concluded that without local estimates of K, RUSLE can considerably overestimate hillslope contribution to the sediment budget. While adjustments of the original nomograph equation to fit Australian soils have been advocated by several authors (Loch and Rosewell 1992; Loch *et al.* 1998; Silburn 2011), few studies in Australia have directly measured field soil erodibility of grazing lands.

Process-based erosion models, such as the Water Erosion Prediction Project (WEPP) can be calibrated on an event basis with runoff and soil loss measurements from simulated rainfall events. In contrast to the empirical RUSLE model, the WEPP incorporates a series of hydrological and erosion processes related to soil, runoff, infiltration, plant growth, hydraulics, management, and climate (Nearing *et al.* 1989). The WEPP also describes detachment and deposition processes, shear stress, and sediment transport capacity, which vary in space and time (Foster *et al.* 1995). The WEPP used simulated rainfall data to develop predictive equations for interrill erosion, rill erosion, and critical sheer stress based on soil properties (Elliot and Flanagan 2023).

The suitability of WEPP has been validated for selected sites in New South Wales and southern Queensland (Yu *et al.* 2000; Yu and Rosewell 2001). Both WEPP and RUSLE share the same ultimate output to provide an estimate of mean annual soil loss. It is therefore reasoned that an appropriately calibrated WEPP could be used to predict long-term mean annual soil loss, which in turn could be used to derive the soil erodibility factor for the RUSLE through rearrangement of

the RUSLE equation. This will provide a low-cost, measurement-based alternative to estimate the K for RUSLE to assist in the modelling of GBR catchment sediment sources, and to better guide sediment reduction strategies to meet Reef 2050 WQIP targets.

This study aims to combine the two erosion prediction models, WEPP and RUSLE, to define and estimate a K-factor based on observed erosion data from plot-scale, simulated rainfall and runoff to determine alternative erodibility estimates to improve hillslope erosion modelling in the GBR catchments. We will estimate K by substituting the WEPP-predicted average annual soil loss (A) into the RUSLE equation for six soils located on grazed land in the Fitzroy Basin of Queensland, Australia. We will compare these estimates against pre-existing K values calculated with the original and modified nomograph equations.

Methods

The experimental design is shown in Fig. 1 where annual soil loss from the RUSLE (A) was replaced with a WEPP-generated annual soil loss (A_{WEPP}).

This is essentially a two-part process where WEPP was calibrated based on field measurements and used to calculate event hydrological and sediment data, which were input into WEPP to estimate erodibility parameters (K_i). The calibrated parameter values, along with site-derived long-term climate data, were then used to run a simulated climate scenario for a 100-year period to generate a long-term mean annual soil loss resulting in A_{WEPP}. The A_{WEPP} was then used to compute K_{WEPP} using RUSLE (Fig. 2), where A_{WEPP} was substituted for A in Eqn 4.

Site description

Data were collected from two beef cattle grazing properties within the semi-arid Fitzroy Basin: (1) the Brigalow Catchment Study and surrounds in 2017; and (2) Belmont Research Station in 2021 (Fig. 3).

The Brigalow Catchment Study is located near Theodore in central Queensland (24.81°S, 149.80°E, GDA 94) and was described in detail in Cowie *et al.* (2007). It is located in the Dawson River catchment of the Fitzroy Basin, and the climate is semi-arid to subtropical with a mean annual rainfall of 650 mm. Average maximum monthly temperature in summer is 33°C (December–February) and average minimum temperature in winter of 6.5°C (June–August). Four sites were investigated in this location: two Brown Sodosols and two Black Vertosols. Sites with similar soil types showed enough soil texture variation to be considered separately as Vertosol$_1$ and Vertosol$_2$, and Sodosol$_1$ and Sodosol$_2$. These soil types are representative of 56% of the Fitzroy Basin with the land use of grazing: Vertosols (28%) and Sodosols (28%) (Roots 2016; Elledge and Thornton 2017). The two Sodosols in this study have a low average exchangeable sodium percentage (ESP) in the 0–0.1 m depth of 0.4% for Sodosol$_1$ and 4.5% for Sodosol$_2$. Vertosol$_1$ showed ESP at the depth of 0–0.1 m was 6.8%, indicating marginal sodicity (Hazelton and Murphy 2007).

Belmont Research Station is located approximately 37 km north of Rockhampton adjacent to the Fitzroy River in the Fitzroy catchment of the Fitzroy Basin (23.23°S, 150.39°E GDA 94) and was described in detail in Coventry and Murtha (1976). The station is located close to the Tropic of Capricorn, and the climate is sub-tropical with a mean annual rainfall of 800 mm. Average maximum monthly temperature in summer is 32°C (December–February), and average minimum monthly temperature in winter is 9°C (June–August). Two sites and soil
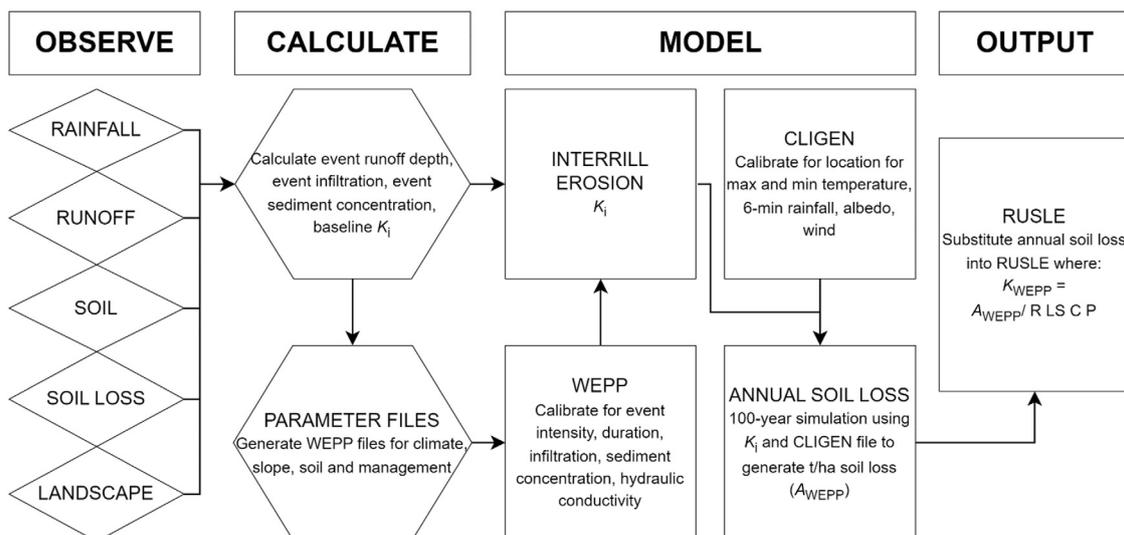

Fig. 1. Conceptual framework to determine A_{WEPP} and K_{WEPP} and from field-measured erosion data.
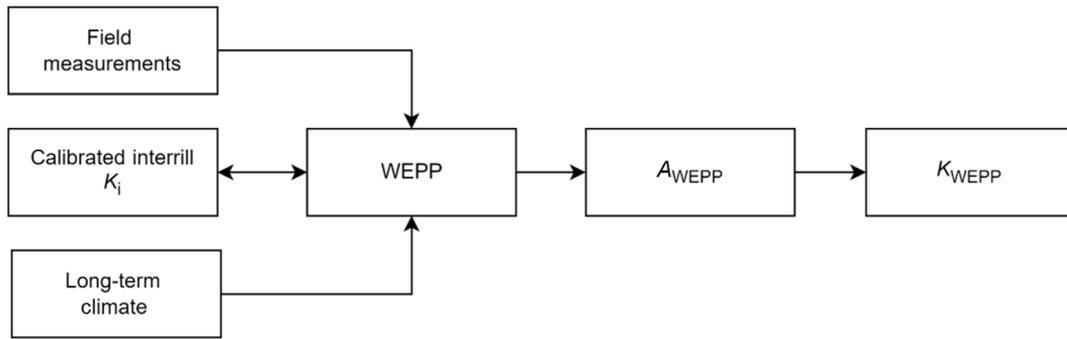
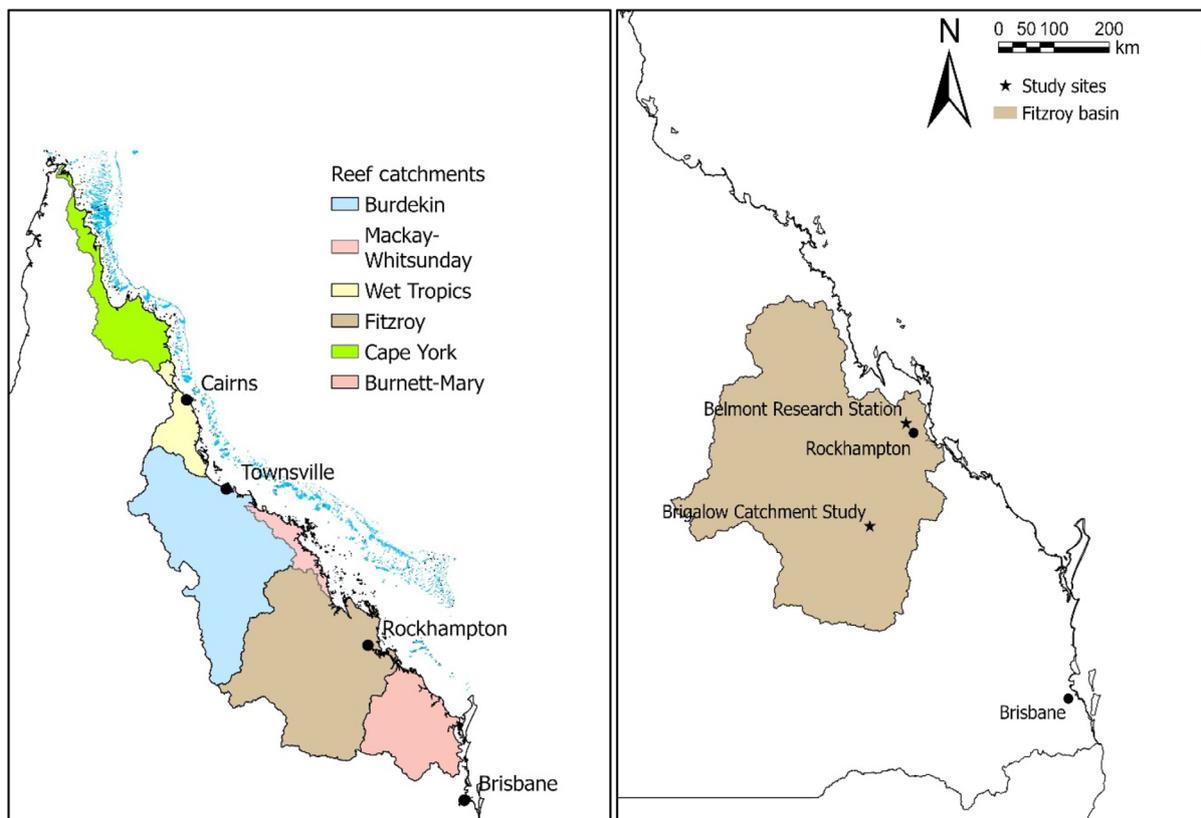**Fig. 2.** Simplified concept to determine K_{WEPP}.

Fig. 3. Relative location of the Fitzroy Basin in the Great Barrier Reef catchments, Queensland, Australia (left), and study locations within the Fitzroy Basin (right).

types were investigated in this location including a Leptic Tenosol and a Black Dermosol. These soils are collectively representative of 14% of the Fitzroy Basin under grazing: Dermosols (11%) and Tenosols (3%) (Roots 2016).

Treatments

All soils were studied *in situ* to preserve soil consolidation and organic bonds. The plot surface was prepared as an artificially scalded surface to represent consolidated bare ground by mechanically removing the surface organic A1 horizon to expose the hard-set layer beneath and was left to weather and reconsolidate for up to three months. Use of artificial scalds in place of natural scalds was validated in Loch (2015a) and Loch (2017).

The Vertosol and Sodosol soils had an additional natural scald treatment as well as an artificial scald. This treatment identified bare surfaces which had been weathered *in situ* for at least a year, eroding the remnant A surface horizon to a hard-set layer.

Rainfall simulation, runoff generation, and water quality sampling

Plots were bunded by inserting flat galvanised panels approximately 0.2 m high into the ground to a depth of at least 0.05 m. The plots sizes on the Brigalow Catchment Study were 2.0 m × 1.5 m (3 m^2) and those on Belmont Research Station were 2.0 m long by 1.4 m wide (2.8 m^2). All plots were on slope gradients less than 5%. A triangular galvanised flume narrowing into a collection pipe was placed downslope to collect runoff water. The difference in plot sizes was due to the width of the flume available at the time of the experiment. Potable water, or rainwater, with low electrical conductivity (<200 mS m^{-1}) was applied using the rainfall simulator described in Loch *et al.* (2001) at a target intensity of 80 mm h^{-1} with VeeJet 80100 nozzles. The target intensity was selected to meet a one in 20 year rainfall event as determined by the Bureau of Meteorology Average Recurrence Interval data (Bureau of Meteorology 2022). Runoff was collected in 1-L sample bottles at the commencement of steady state runoff, with water quality samples taken every 3–5 min for 20–30 min. Each runoff sample was timed during collection and weighed after the event to calculate a runoff rate (mm h^{-1}). Water quality samples were analysed at the Department of Environment, Tourism, Science and Innovation Chemistry Centre for total solids and volatile solids (method 2540B) and total suspended solids (method 2540D). Water analysis methods were based on APHA AWWA (2012). Each site was visited once with three rainfall simulation repetitions conducted on each soil type and treatment. A comparison of the observed hydrograph and sediment output per plot and treatment was conducted to validate each plot.

Soil sampling

Surface soil samples (0–0.1 m) were collected pre- and post-rainfall simulation events to determine soil moisture conditions. Soil cores were analysed for air dried moisture content, cation exchange capacity, total organic carbon, bulk density, and particle size fractions of clay, silt, and sand using the methods of Rayment and Lyons (2011). Percent of undispersed particles less than 0.125 mm in the rainfall wet soil surface (P$_{125}$) was determined according to Loch and Rosewell (1992) (Table 1).

Soil erosion modelling: WEPP

The WEPP can be used for multiple-year, long-term simulations as well as single event simulations. For this study, WEPP v2012.8 was run for each experimental plot using the single storm option and plot-specific input files accounting for soil properties, slope geometry (length and slope), ground cover, and rainfall intensity. Soil hydraulic conductivity (K_e) was calibrated in the soil file until the WEPP single storm infiltration rate equalled the observed steady state infiltration rate per plot. The K_e was averaged per soil type and treatment. Steady state was defined as the period when the hydrograph and sediment concentration are constant over time, as distinct from noticeable fluctuations at the beginning of the runoff event. Interrill erodibility in the individual WEPP soil files was adjusted to match modelled sediment loss per unit width (kg m^{-1}) against observations. The observed soil loss was related to observed steady state concentration (g L^{-1}) and converted to kg m^{-1} via:

$$E = 0.001cLQ \tag{1}$$

where E is the soil loss per unit width (kg m^{-1}), c is the steady state concentration (g L^{-1}), L is slope length (m), and Q is total runoff depth (mm).

Long-term climate data for Belmont Research Station covered a period of 45 years based on Bureau of Meteorology (data from the Rockhampton station. Climate data for the Brigalow Catchment Study were collected onsite, providing a 43-year dataset. These climate data were used to generate the 100-year weather simulation using CLIGEN v5.3.

Soil erosion modelling: RUSLE

The RUSLE equation (Renard *et al.* 1997) follows:

$$A = RKLSCP \tag{2}$$

Table 1. Surface soil properties and selected chemical analysis of the six soils observed in this study (0–0.1 m depth).

Soil	Plot area (m^2)	Slope (%)	Clay (<0.002 mm) (%)	Silt (0.02–0.002 mm) (%)	Fine sand (0.2–0.02 mm) (%)	Coarse sand (>0.2 mm) (%)	P$_{125}$ (<0.125 mm) (%)	Estimated wet density (Mg m^{-3})	Organic carbon (%)	Cation exchange capacity (cmol$_c$ kg^{-1})	Bulk density (g cm^{-3})
Dermosol	2.8	3.5	30.1	24.4	22.1	25.0	63.7	1.68	1.52	18	1.31
Sodosol$_1$	3.0	2.8	10.7	4.2	39.9	49.4	32.9	2.31	0.96	13	1.58
Sodosol$_2$	3.0	3.3	20.9	7.4	39.2	36.3	52.6	2.02	0.73	10	1.65
Tenosol	2.8	5.0	19.8	31.2	23.6	30.1	64.8	1.70	1.25	13	1.42
Vertosol$_1$	3.0	2.2	34.8	10.7	35.6	27.7	49.1	1.82	0.85	25	1.54
Vertosol$_2$	3.0	2.3	26.1	7.5	37.8	29.8	48.2	2.00	1.53	18	1.62

P$_{125}$ is the percentage of undispersed soil particles <0.125 mm in the soil surface after rainfall wetting (Loch and Rosewell 1992).

where A is annual soil loss (t ha^{-1} year^{-1}); R is the annual rainfall erosivity (MJ mm ha^{-1} h^{-1} year^{-1}); K is soil erodibility (t h MJ^{-1} mm^{-1}); L and S are the slope length and slope steepness factors, collectively referred to as the LS factor; C is the crop cover and management factor; and P is practice factor which includes contouring, terracing, and other conservation practices. The LS, C, and P are unitless, describing the deviation from a USLE unit plot. The USLE unit plot is defined as a length of 22.1 m with a 9% slope, tilled up and down the slope, and retained as bare fallow conditions. Units here are expressed as SI units converted from the original imperial measure by Foster *et al.* (1981).

For erodibility, the equation for the USLE nomograph K (K_{nom}) in SI units (t h MJ^{-1} mm^{-1}) for soils with less than 70% silt used by Rosewell and Edwards (1988) in Australian studies follows:

$$K = 2.77\,M^{1.14}(10^{-7})(12 - \text{OM}) + 4.28(10^{-3})(\text{SS} - 2) \\ + 3.29(10^{-3})(\text{PP} - 3) \tag{3}$$

where M = (percent silt + percent very fine sand) (100 − percent clay), OM is organic matter (%), SS is the soil structure code, and PP is the profile permeability class. The particle size class defined for silt plus very fine sand is 0.002–0.1 mm and for clay is the less than 0.002 mm fraction. The Loch *et al.* (1998) nomograph modification (K_m) adjusts the definition of M, where M is replaced by 100(P$_{125}$), P$_{125}$ is particle size distribution less than 0.125 mm, and by using estimated or measured wet sediment density from Loch and Rosewell (1992). Eqn 3 and the K_m modification are used in the GBR Dynamic SedNet and were therefore selected for this study.

In the RUSLE, the ratio of the average soil loss (A) per unit of rainfall erosivity (R) for a unit plot under continuous fallow is represented as $K = A/R$. In this experiment, LS and C were used to represent a low rill/interrill relationship on a bare plot, and the resulting K-factor follows:

$$K = A/(RLSCP) \tag{4}$$

The soil loss A was estimated from WEPP by running 100-year simulations based on the plot-calibrated effective hydraulic conductivity (mm h^{-1}) and interrill erodibility (kg s m^{-4}) per soil type. The R was obtained from table 2 in Yu (1998) using Station 39083 for Belmont Research Station, and Station 39090 for the Brigalow Catchment Study. A 100-year simulation was run in WEPP to model the annual average soil loss output (A_{WEPP}) at a slope length of 22 m as reflective of the initial USLE plot at a 9% gradient. The C-factor parameters were adapted from the cover factor equation in Rosewell (1993) where 0.45 is for bare ground (Silburn 2011). The P-factor was set to 1 as it was not applicable here.

Data analysis

RStudio version 4.2.2 was used to analyse and generate graphics of the data using *base R*, elements of *tidyverse* including *dplyr* (Wickham *et al.* 2023) and *ggplot* (Wickham 2016) packages. Other packages used to explore and analyse the data include *broom* (Robinson *et al.* 2023), *multcomp* (Hothorn *et al.* 2023), and *car* (Fox and Weisberg 2019). *Post hoc* Bonferroni adjusted multiple comparison analysis was used to identify group differences. Results in Tables 2 and 3 are presented with alphabetical letters where groups not sharing the same letter differ at the 5% level. Performance measures used to compare WEPP models with observations include R^2, PBIAS, and NSE (Samantaray *et al.* 2024, 2025). They are defined below, where OBS is observed data, $\overline{\text{OBS}}$ is the mean, and SIM is the WEPP simulated results. The R^2 is based on the formula used in *tidyverse* where SS is sum of squares for residuals (res) and totals (tot).

$$R^2 = 1 - \frac{\text{SS}_{res}}{\text{SS}_{tot}} \tag{5}$$

$$\text{PBIAS} = \frac{\sum \text{SIM}_i - \sum \text{OBS}_i}{\sum \text{OBS}_i} \times 100\% \tag{6}$$

$$\text{NSE} = 1 - \frac{\sum (\text{OBS}_i - \text{SIM}_i)^2}{\sum (\text{OBS}_i - \overline{\text{OBS}}_i)^2} \tag{7}$$

Results

Measured data for the rainfall simulation plots

There were no significant differences for any parameter between natural and artificial scalds on Vertosols and Sodosols ($P > 0.7$). Based on this result, hydrograph properties for the natural and artificial scalds on these two soil types were combined (Table 2). Minimal to no rilling was observed in the 2-m plots. Runoff started 1.5–4.3 min after the commencement of simulated rainfall, and soils with a higher clay and silt content began runoff earlier. A similar trend was noted for steady state infiltration (mm h^{-1}) where coarser textured soils maintained a higher infiltration rate. Sediment loss peaked early in the runoff events (0–5 min) and gradually declined with time for most plots. Most plots had an observable steady state, with soil loss remaining relatively constant across most times steps except for the Tenosol, which consistently declined for each time step, as did Sodosol$_2$ to a lesser degree. Steady state observations are summarised for infiltration (mm h^{-1}), sediment concentration (g L^{-1}), sediment generation (g min^{-1}), and runoff rate (mm h^{-1}) in Table 2.

The Dermosol, while higher in clay content (30%) and with a low estimated wet density (1.68 Mg m^{-3}), had a high steady state infiltration rate, exceeded only by the sandy loam of the Sodosol$_1$. The Dermosol and Tenosol runoff depths were the

Table 2. Observed steady state mean hydrological properties.

Soil	n plots	Slope (%)	Time to runoff (min)	Rainfall intensity (mm h^{-1})	Rainfall duration (min)	Runoff rate (mm h^{-1})	Runoff depth (mm)	Infiltration rate (mm h^{-1})	Sediment concentration (g L^{-1})	Sediment discharge (g min^{-1})
Dermosol	2	3.5	1.5[c]	84	33	72.3(0.3)[b]	44.2(0.4)[d]	11.5(1.2)[ab]	5.4(0.2)[b]	18.0(0.7)[b]
Sodosol$_1$	2	2.8	4.0[a]	77	20	54.0(2.4)[c]	31.8(0.3)[c]	16.2(0.9)[a]	13.1(1.3)[a]	36.5(2.4)[ab]
Sodosol$_2$	6	3.3	2.0[c]	88	20	83.4(0.6)[a]	26.3(0.4)[b]	5.1(0.3)[b]	8.6(0.4)[ab]	36.4(1.9)[ab]
Tenosol	3	5.0	1.7[c]	82	33	81.1(0.9)[ab]	43.3(0.4)[d]	6.4(0.6)[b]	12.8(0.2)[a]	51.4(2.0)[a]
Vertosol$_1$	3	2.2	2.7[b]	91	20	79.3(0.7)[ab]	32.4(0.2)[c]	8.5(0.8)[ab]	10.1(0.1)[ab]	40.9(0.2)[ab]
Vertosol$_2$	6	2.3	3.7[a]	83	20	75.1(0.5)[b]	28.4(0.3)[a]	8.2(0.4)[b]	7.4(0.2)[b]	25.7(0.9)[b]

Means followed by the same letter within columns are not statistically different. Standard errors are given in parentheses.

Table 3. WEPP parameters where K_{nom} is calculated nomograph K, K_m is calculated from Loch *et al.* (1998), and K_{WEPP} is erodibility calculated from modelled A in WEPP.

Soil	n plots	RUSLE R-factor (Yu 1998)	WEPP mean infiltration (K_e) (mm h^{-1})	Calculated K_i using Elliot *et al.* (1989) (kg s m^{-4} × 10^6)	WEPP mean interrill erosion rate (K_i) (kg s m^{-4} × 10^6)	WEPP annual soil loss (t ha^{-1})	C-factor	K_{WEPP} initial (t h MJ^{-1} mm^{-1})	Average K_{nom} (t h MJ^{-1} mm^{-1})	Average K_m (t h MJ^{-1} mm^{-1})	Average K_{WEPP} (t h MJ^{-1} mm^{-1})
Dermosol	2	3152	2.3[b]	0.57[c]	0.97[b]	20.6	0.39	0.007	0.032	0.045	0.017
Sodosol$_1$	2	2683	20.0[a]	1.61[ab]	1.57[ab]	31.6	0.45	0.012	0.029	0.032	0.026
Sodosol$_2$	6	2683	2.4[b]	1.12[bc]	1.12[b]	46.4	0.45	0.017	0.034	0.036	0.038
Tenosol	3	3152	0.7[b]	1.90[a]	2.26[a]	39.4	0.45	0.013	0.040	0.048	0.028
Vertosol$_1$	3	2683	5.9[b]	1.31[abc]	1.06[b]	21.4	0.45	0.008	0.035	0.042	0.018
Vertosol$_2$	6	2683	5.3[b]	0.97[bc]	0.96[b]	21.0	0.45	0.008	0.033	0.034	0.017

Means followed by the same letter within columns are not statistically different.

highest at 44 and 43 mm, respectively. Runoff depth results are caveated by the differences in rainfall duration, with most sites receiving 20 min of rain except for the Dermosol and Tenosol with 33 min. All plots receiving ~20 min of rain had runoff depths of 26–32 mm. Runoff initiated earliest at 1.5 min on the Dermosol. The Dermosol had the lowest sediment generation (18.0 g min^{-1}) and concentration (5.4 g L^{-1}) of all soils investigated. The runoff rate from all Dermosol plots averaged 72 mm h^{-1} with an average rainfall intensity of 84 mm h^{-1}.

The Sodosol$_1$ was a loamy sand textured soil with a low clay and silt content (9.9% and 3.4%, respectively), and time to runoff from rain initiation was the slowest at 4.0 min. Infiltration was the highest (16 mm h^{-1}) as was sediment concentration (13.1 g L^{-1}). Sodosol$_1$ had the lowest runoff rate (54 mm h^{-1}). In contrast, Sodosol$_2$, which had a higher clay and silt content (20.9% and 7.4%, respectively), had the lowest infiltration rate (5.1 mm h^{-1}), the highest runoff rate (83.4 mm h^{-1}), and the lowest runoff depth for plots receiving 20 min of rain (26.3 mm). The Tenosol had the highest sediment rate (51.4 g min^{-1}) and second highest sediment concentration (12.8 g L^{-1}).

The Vertosol$_1$ showed differences to Vertosol$_2$, with the higher mean rainfall intensity (91 mm h^{-1}) resulting in a higher runoff rate, depth, sediment concentration, and sediment rate than Vertosol$_2$. Soil particle size fractions were similar between the two Vertosols; however, Vertosol$_2$ had higher organic carbon (1.53%) than Vertosol$_1$ (0.85%). Sediment concentration and sediment generation for Vertosol$_2$ were low.

All steady state factors of infiltration (mm h^{-1}), runoff rate (mm h^{-1}), sediment concentration (g L^{-1}), and sediment generation rate (g min^{-1}) significantly differed between soil types ($P < 0.002$). Significant differences were observed between most soil types for runoff depth (mm), primarily influenced by differing rainfall durations and therefore the P-value is not shown here.

Infiltration rates decreased as time increased (Fig. 4), whereas runoff rates increased (Fig. 5). For Sodosol$_1$ and to a limited degree Vertosol$_2$, sediment transport increased initially before decreasing as time passed (Fig. 6). Steady state sediment generation was achieved for most soils, with the Tenosol and Sodosol$_1$ the exceptions. For the Tenosol, and to a lesser degree Sodosol$_1$, sediment generation was still declining at the end of the rainfall event (Fig. 6).

WEPP modelling outputs

Each plot repetition per soil and scald treatment were validated using comparisons of the observed plot hydrograph
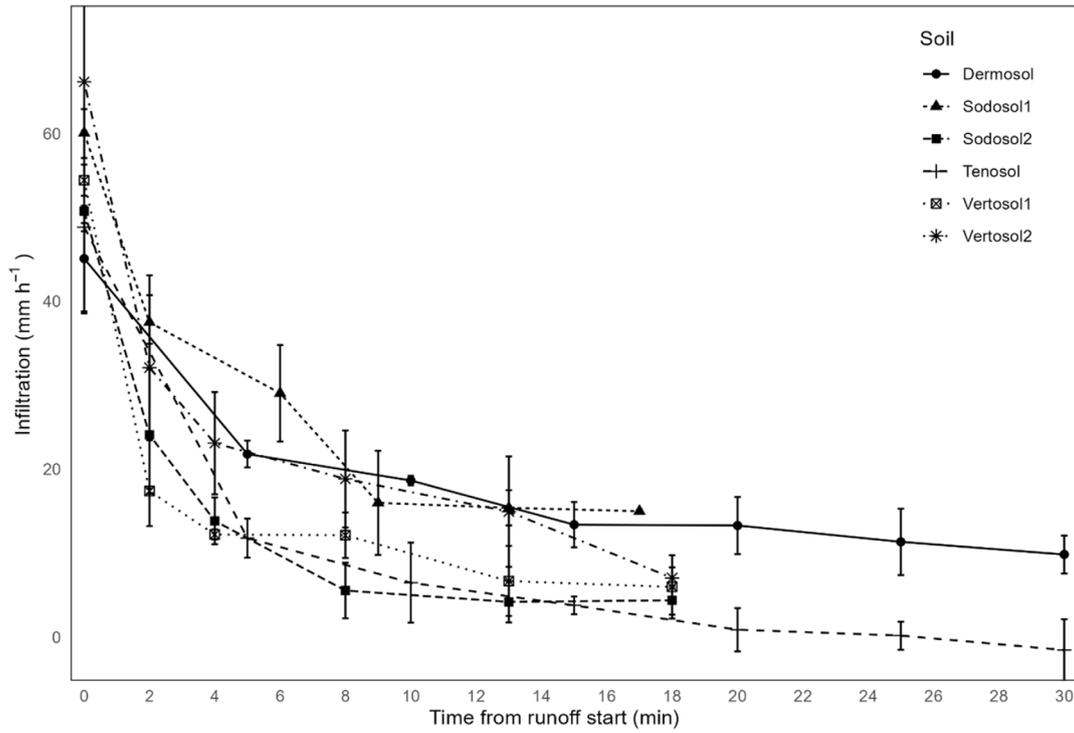
Fig. 4. Observed infiltration rate across the runoff events per soil type.
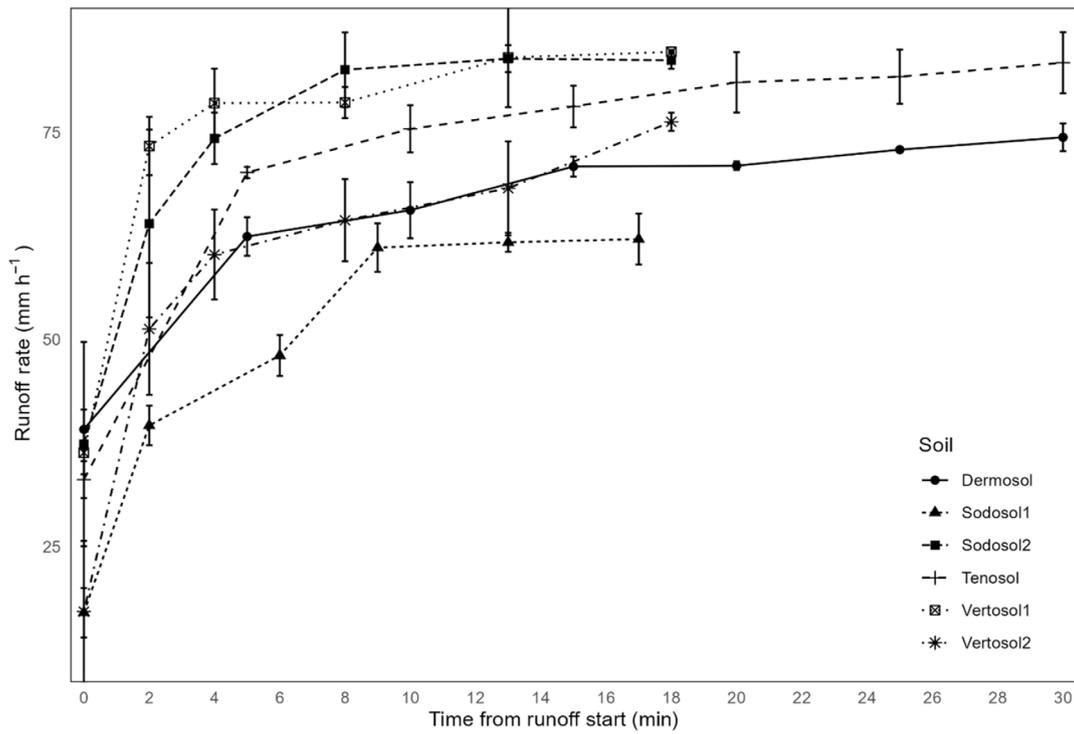

Fig. 5. Observed runoff rate across the runoff events per soil type.

and sediment output, then combined by treatment and soil type. Artificial scalds were the only treatment observed on

the Dermosol and Tenosol. Data modelled in WEPP for infiltration rate (K_e, mm h^{-1}), mean interrill erodibility
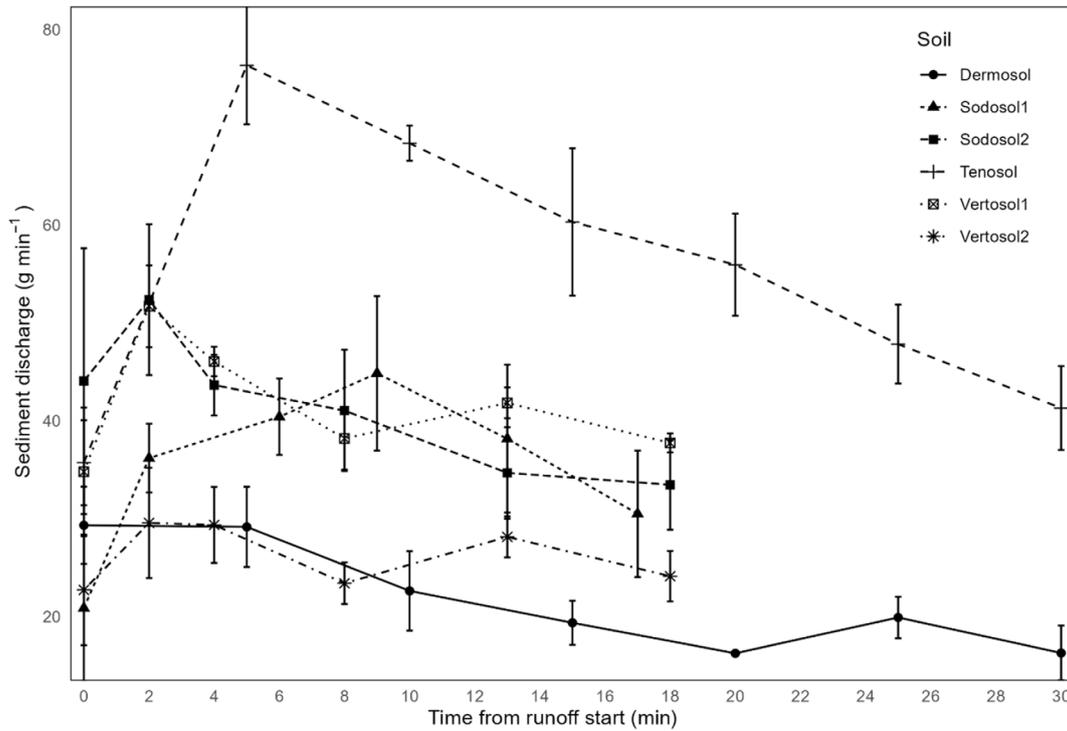
Fig. 6. Observed sediment discharge across the runoff events per soil type.

$(K_i, \times 10^6 \text{ kg s m}^{-4})$, annual soil loss (t ha^{-1}), and the cover factor are shown in Table 3 alongside modelled K_{WEPP} (erodibility) parameters (t h MJ^{-1} mm^{-1}). Initial K_{WEPP} was the result of $K = A/R$ prior to applying Eqn 4.

Total observed soil loss (t ha^{-1}) from each simulated rainfall event was compared to the WEPP modelled soil loss on a plot-by-plot basis to assess WEPP outputs (Fig. 7). The
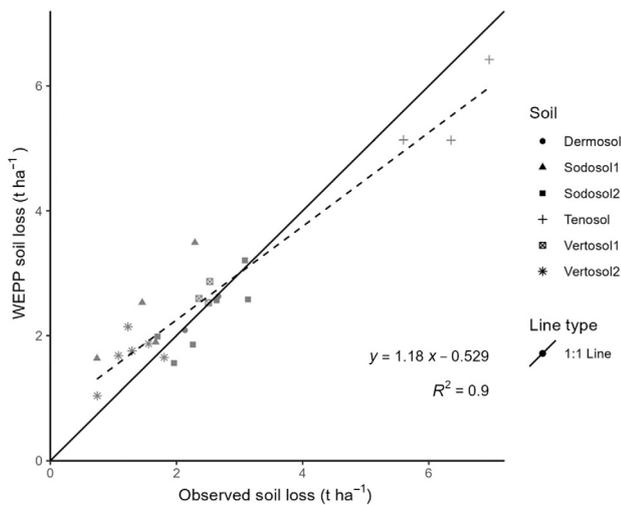

Fig. 7. Comparison of observed plot-scale soil loss for each simulated rainfall event against WEPP-modelled soil loss per plot ($R^2 = 0.9$, PBIAS = 1.9%, NSE = 0.87, $n = 24$).

model fit ($R^2 = 0.9$, PBIAS = 1.9%, NSE = 0.87) indicates good confidence that the plot calibrations are acceptable.

The WEPP mean infiltration (K_e) was calibrated per plot then averaged across each repetition per scald treatment and soil type. Where a plot was not valid, it was dropped from analysis. Hydrological anomalies observed for Vertosol$_1$ reduced a potential six plots to three, and on Sodosol$_1$, six plots were reduced to two. The Dermosol included two plots of a potential three. Modelled K_e appeared to follow the same observed steady state infiltration rate trends as field observations, where the sandier textured Sodosol$_1$ showed considerably higher K_e (20.0 mm h^{-1}). The Tenosol had the lowest K_e of 0.7 mm h^{-1}. The K_e significantly differed between soil types ($P < 0.001$) (Table 3).

Interrill erosion (K_i) parameters significantly differed between soil types ($P < 0.001$). Calculated K_i based on Elliot et al. (1989) was tested against modelled WEPP K_i and showed good agreement for most soils and both approaches, with derived K_i not significantly different ($P = 0.9$). The highest K_i value was for the Tenosol of 2.26×10^6 kg s m^{-4} and the lowest was 0.96×10^6 kg s m^{-4} for Vertosol$_2$.

The K_m (modified nomograph) across all soils ranged between 0.032 and 0.048 t h MJ^{-1} mm^{-1}, indicating moderate erodibility for each soil studied. For K_{nom}, the range was 0.034–0.040, also indicating moderate erodibility for all soils. When using A_{WEPP} to substitute into RUSLE, the resultant K_{WEPP} range for most soils was 0.017–0.028, indicating low erodibility class, with Sodosol$_2$ the only moderately erodible

soil (0.038), which was also the highest K-factor in this study for K_{WEPP}. For Sodosol$_2$, K_m is within 8% of the modelled K_{WEPP}. The Dermosol and Vertosol$_1$ showed the lowest K_{WEPP} erodibility (0.017, low erodibility), noting the Dermosol had enough rock cover on the bare surface to reduce the C-factor to 0.39.

The K_{WEPP} was up to 63% lower than K_m for all soils, except Sodosol$_2$ where it was 8% higher (Fig. 8). Vertosol$_1$ and Vertosol$_2$ were 49% and 58% lower, respectively, when comparing K_{WEPP} to K_m. Both nomograph-calculated K-factors (K_{nom} and K_m) indicated moderate to high erodibility for Vertosols (range 0.033–0.042). Sodosol$_1$ was 17% lower for K_{WEPP} than K_m. The Tenosol was 42% lower for K_{WEPP} and the Dermosol 63% lower for K_{WEPP}.

Discussion

WEPP/RUSLE integration

Limited Australian studies have investigated estimating soil erodibility using WEPP as a substitute for RUSLE, with the exception of developmental work undertaken between 2015 and 2017 as part of the Paddock to Reef Integrated Monitoring and Modelling Program reported in Loch (2017). In Loch (2017), data observed for the Brigalow Catchment Study were modelled using shorter slope length than this study (11 m by 4%) and an extended version of WEPP not publicly available. Loch (2017) used a modified version of WEPP v2012.8, which allows up to 10 different soil size classes as opposed to the three classes used in the publicly available WEPP v2012.8 version, introducing a variability this study cannot account

for. No specific methodology or WEPP soil, management, and climate files used to develop K-factors in Loch (2017) are available and were subsequently independently developed for this study.

Internationally, there have been several studies comparing erodibility from the RUSLE and the WEPP. Nearing and Nicks (1998) compared erodibility of 45 soils in the USA and noted that direct comparison between the RUSLE and WEPP erodibilities is complicated by the inclusion of three erodibility-related parameters within WEPP (interrill, rill, and critical shear stress); however, there was no statistical bias between the two model outputs.

Laboratory experiments by Deviren Saygin *et al.* (2018) investigated the combination of WEPP and RUSLE to estimate the USLE erodibility factor. Deviren Saygin *et al.* (2018) used laboratory-based rainfall simulation plots to separately calculate interrill and rill parameters to input into WEPP to output an annual soil loss over a 100-year simulation to calculate K. Modelled K in their study was significantly lower than the empirical RUSLE equations for K. The approach by Deviren Saygin *et al.* (2018) differed to this study by applying a monthly tillage in the WEPP model, where this study considers a permanent consolidated surface as found on grazed hillslopes. That study also used soil transferred to laboratory plots to apply simulated rain, which may not represent the consolidation and organic bonds of *in situ* soil.

Interrill erosion

As ascertained in Yu and Rosewell (2001), WEPP is a complex process-based model comprising a significant number of
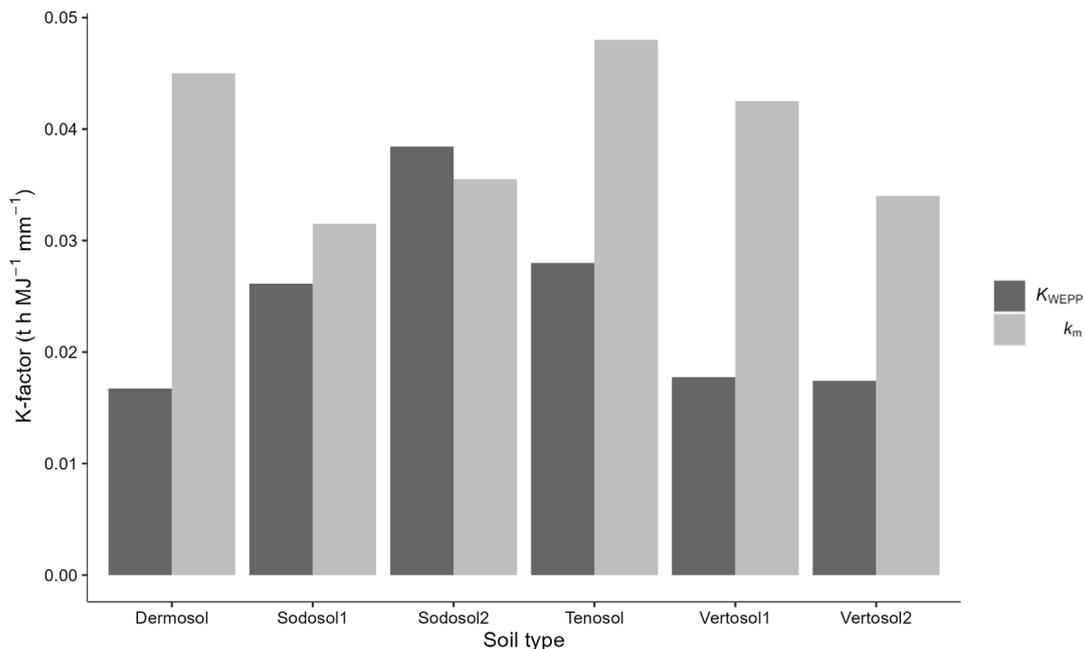

Fig. 8. Comparison of K_{WEPP} and K_m erodibility parameters.

initial parameters and sub-routines. Every effort was made to set initial parameters as accurately as possible, based on observed site data. All erosion observed was interrill erosion, as expected for plots with short lengths and low slopes. While this study attempted to isolate interrill erosion specifically as the driver for soil loss, the WEPP model still considers rill formation when scaling the plot data to a 22 m slope length, contributing some rilling to the sediment transport output.

Interrill erosion processes are dominated by raindrop impact and subsequent sediment transport of overland flow (Kinnell 2005; Zhang *et al.* 2020). When raindrop detachment is greater than the transport capacity of overland flow, the system is transport-limited; if flow is greater than the ability of raindrops to detach soil particles, the system is detachment-limited (Zhang *et al.* 2020). Both transport- and detachment-limited processes were observed in this study, where initial sediment loss peaked early where raindrop-impact detached soil, declining with time as entrainable sediment supply reduced, and the ability for raindrop impact to disperse particles was less effective as depth of flow increased. In this study, low slopes inhibited flow transport and are representative of the low gradient of grazed lands in the Fitzroy Basin.

Assessing sediment output using WEPP has been tested in several studies. In Australia, Yu and Rosewell (2001) validated soil loss prediction on bare fallow plots, providing confidence in the substitution of A in the RUSLE for soil loss prediction on similarly prepared plots to this study. The WEPP has been successfully applied in various USA conditions to predict sediment yield (Ghidey and Alberts 1996; Ahmadi *et al.* 2011). Accurately representing the hydrology and climate variables assists in better predictability of soil loss (Laflen *et al.* 1994), and with known parameters of the rainfall event applied to the plot, measured hydrology, and an appropriately-generated CLIGEN file, the calibration to simulate soil loss was applied with the best available data.

Modelled *K*-factor

In soils with similar totals of clay and silt (approximately 50% combined total for the Dermosol, Tenosol, and Vertosol$_1$), observed sediment loss, infiltration, and sediment transport varied significantly. These differences may be influenced by variations in the soil properties of cation exchange capacity and organic matter as captured in the nomograph; however, the variation between the nomograph equations (K_m and K_{nom}) and the modelled K_{WEPP} are worthy of discussion. The modified nomograph (K_m), limited to soil properties, indicates that all soils are moderately erodible, with the Tenosol, Dermosol, and Vertosol$_1$ at the high end of the moderately erodible scale. The K_{nom} follows a similar trend; however, K_{WEPP} disagrees with K_m and K_{nom}.

As an example, the Tenosol had the highest K_m (0.048) and K_{nom} (0.040) but was in the low erodibility class for K_{WEPP} (0.028), despite having the highest sediment discharge rate and interrill erosion parameter. Literature shows that

Tenosols can vary from low to high erodibility. Silburn (2011) provides a measured *K*-factor for an Orthic Tenosol of 0.050 in central Queensland's Nogoa sub-catchment of the Fitzroy Basin, and for a Tenosol in Victoria's LaTrobe catchment, Vigiak *et al.* (2011) provides an estimate of 0.015; however, the Tenosol in Victoria had very low clay (1%) and very high organic matter (12%). The Tenosol in Silburn (2011) had 12% clay and low organic matter of 0.6%. The K_{WEPP} of 0.028 modelled here is a likely representation of *in situ*, undisturbed soil and seems reasonable for a Tenosol, as is K_m of 0.048.

Vertosol$_1$ and Vertosol$_2$ sites were located within the conservatively grazed catchment of the Brigalow Catchment Study. This catchment has a long-term observed soil loss of 0.26 t ha^{-1} year^{-1} for the years 1984–2010 (Thornton and Elledge 2021). Substituting K_{WEPP} (0.018) into the RUSLE equation as K, and appropriate parameters for the catchment (LS = 0.37; R = 2683; C = 0.013), the soil loss estimate was 0.23 t ha^{-1} year^{-1}. Using the average K_m for Vertosol$_1$ and Vertosol$_2$ (0.038), the predicted soil loss output was 0.49 t ha^{-1} year^{-1}; double the observed data. A *K*-factor developed by Loch (2017) with a similar WEPP and RUSLE integration based on the same observed dataset (K = 0.023) when substituted into RUSLE, also provided a reasonable estimate for this same catchment (0.30 t ha^{-1} year^{-1}) compared to the long-term observed soil loss. An analysis of the Modified USLE (MUSLE) by Tiwari *et al.* (2021) estimated this conservatively grazed catchment had a *K*-factor of 0.059 which was significantly higher than the WEPP-derived K estimates provided here.

Vertosols are prevalent in the Fitzroy Basin (28% of the grazed land use). Vertosols studied by Freebairn *et al.* (1996) at the Darling Downs, Queensland, indicate that Vertosols are highly erodible, specifically on cropping systems. The soils in that study had in excess of 50% clay unlike this study. Sediment loss data from the Freebairn *et al.* (1989) study were evaluated against the USLE, and it was reported that deriving erodibility from the nomograph equation was determined to be unreliable for high clay soils (Freebairn *et al.* 1989). Measured erodibility from the cropping trial by Freebairn *et al.* (1989) was used as a point of validation for modelling erodibility using WEPP and RUSLE by Loch (2015*b*), which showed reasonable agreement, indicating the use of WEPP to derive an annual soil loss average to substitute into RUSLE is feasible.

Aggregate stability is an indicator of susceptibility to erosion (Barthès and Roose 2002; Ben-Hur and Lado 2008), particularly the interaction of clay particles (Lado and Ben-Hur 2004). Vertosols are characterised by high clay content (Isbell and National Committee on Soil and Terrain 2021) and Murphy *et al.* (2013) makes the point that when wet, Vertosols are highly erodible. At the time of the rainfall simulation study on the Vertosols, the soil moisture prior to applying rain was low (~4%). This study showed variation of sediment discharge between the Vertosols modelled (40.9 and 25.7 g min^{-1}). This may be explained by

variations in soil organic carbon (SOC) and clay fractions, with Vertosol$_1$ exhibiting lower SOC (0.85%) and higher clay content (34.8%) than Vertosol$_2$ (1.53% and 26%, respectively). Despite these differences, and variations in cation exchange capacity and soil texture, the K_i modelled for the Vertosols was similar (1.06×10^6 and 0.96×10^6 kg s m^{-4}) and, as a result, so was the K_{WEPP} simulated under the same climactic conditions (K_{WEPP} 0.018 and 0.017).

Using the Sodosol$_1$ dataset, Loch (2017) modelled a 100-year WEPP simulation with a slope length of 11 m by 4% slope with different RUSLE C-factors (0.07 and 0.43) and calculated an average K_{WEPP} of 0.038, which was higher than the modelled output here (K_{WEPP} 0.026) by 32%. For Sodosol$_2$, Loch (2017) derived an average of 0.020 at a LS of 11 m by 4% (C-factor, 0.43). Using a C-factor of 0.45 and a 22 m by 9% LS in this study, Sodosol$_2$ K_{WEPP} was 0.038, which presents a significant difference. On a larger spatial scale (74 m^2), Silburn (2011) calculated a K-factor value of 0.042 for a sandy clay loam, a Brown Sodosol, near Springvale, Queensland, over seven years of observation. This is reasonably comparative to Sodosol$_2$ but 38% higher than Sodosol$_1$. The soil texture properties of Sodosol$_1$ were more similar to the Sodosol studied by Silburn (2011) than Sodosol$_2$. Local variation is therefore relevant and site-specific K-factors would be useful.

One of the few Australian studies where K-factor was measured were on cultivated modified USLE plots (41 m × 2.44 m, 10% slope, 107 m^2) was in Wagga Wagga, New South Wales, where five soils were observed over 4–8 years by Rosewell (1992). That study highlighted the influence of rainfall erosivity variations from year to year on a sandy clay loam (Northcote Principal Profile Form: Dr2.32) similar to Sodosol$_2$, which generated K-factors of 0.003–0.121. This substantial variability indicates that rainfall erosivity variations over time are relevant, highlighting that long periods of *in situ* measurement of soil loss are generally advisable to determine the K-factor. In this study, long-term climate variables were captured in the WEPP modelling using CLIGEN, which were based on a long-term rainfall dataset. The variability of the Sodosols across the above studies makes it difficult to assess an appropriate K-factor; however, considering the sodic influence within the Sodosol soil group, erodibility values have the potential to be significant.

This same study by Rosewell (1992) investigated a possible medium clay Dermosol at Inverell, New South Wales (Northcote Principal Profile Form: Uf6.21) on a bare fallow USLE plot 41 m in length at 10% slope, reporting an average K-factor of 0.017. Annual variations observed there were 0.007–0.061. At the same site, a heavy clay Dermosol (Northcote Principal Profile Form: Uf6.21) had an average K of 0.045 (annual variation 0.018–0.070). The Dermosol in this study showed more soil texture similarities to the heavy clay soil at Inverell. A black Dermosol was investigated by Silburn and Bosomworth (2023) in a cropping scenario, and they determined K_i of 3.9×10^6 kg s m^{-4}, which was significantly higher than K_i estimated in this study

(0.59×10^6 kg s m^{-4}). While both Dermosols had similar clay and silt contents, the loose, unconsolidated condition of the cropping Dermosol resulted in high soil loss under laboratory-simulated rainfall conditions (29.3 t ha^{-1}) compared to *in situ* plots in this study where the consolidated Dermosol soil averaged a soil loss of 2.4 t ha^{-1} (Fig. 7). Consolidation may explain the difference in K_i, as well as the cropping Dermosol having considerably higher fine sand content (45%) than the grazing Dermosol of this study (22%). Considering that Dermosols can have high clay or loamy textures, the response to detachment and transport of sediment will be soil and site specific. The K-factor modelled in this study of 0.017 is therefore reasonable and representative of a cohesive surface.

Particle size

The study data suggest cohesiveness, consolidation, and aggregation present in the soil surface influence detachment compared to an erodibility nomograph equation which expects some surface disturbance (e.g. cultivation). Soil aggregate strength, specifically consolidation of *in situ* soils, influences particle detachment and erodibility, and is integral in erosion resistance (Bryan 2000; Ben-Hur and Lado 2008; Thomaz and Fidalski 2020; Xia *et al.* 2022; Liu *et al.* 2023). While particle size is an influential factor, as mentioned by several authors, a variety of other factors increase or decrease resistance to detachment, surface sealing, and entrainment (Alberts *et al.* 1987; Ben-Hur and Agassi 1997; Ben-Hur and Wakindiki 2004). This was observed here where differing soil property combinations modelled different A_{WEPP} and K_{WEPP} values. The Dermosol and Tenosol had high silt percentages of 24% and 31%, respectively, compared to the other soils; however, these soils exhibited vastly different K_i and K_{WEPP} values. The Dermosol had rock content on the surface and within the soil profile, which may have assisted with infiltration pathways despite a high runoff depth, and thereby inhibited sediment export. Sodosol$_1$ had low clay and silt (~15% combined) and a higher presence of sand fractions, resulting in a high K_i, whereas Sodosol$_2$ had a lower K_i yet had the highest K_{WEPP} of all soils studied. Sodosol$_2$ had the lowest organic carbon percentage (0.73%), generally an indicator of weaker aggregate strength; however, this alone does not explain the high K_{WEPP}, as Vertosol$_2$ and Sodosol$_1$ had organic carbon less than 1% and modelled low K_{WEPP} erodibility.

K_{WEPP} implications

With the exception of Sodosol$_2$, K_{WEPP} was consistently lower than K_m and K_{nom}. This suggests that hillslope RUSLE modelling in Dynamic SedNet, which currently implements K_m, may be overestimating hillslope sediment loss. This has been highlighted in this study where observed and modelled K-factors showed a substantially lower K_{WEPP} than both

erodibility nomograph equations. The nomograph, and indeed the RUSLE, assume a disturbed surface soil not always present on grazed hillslopes when not considering animal traffic itself. Reichert *et al.* (2001) indicates that increased soil aggregation, consolidation, and surface residue reduce erosion. These conditions are analogous to a grazed hillslope as studied here, so a similar response would be expected.

For the soils considered in this study, which together comprise approximately 70% of the area grazed in the Fitzroy catchment, hillslope erosion could be overestimated, specifically on bare, consolidated ground. With increased ground cover on consolidated soils, the expected soil loss response from hillslope grazing would decrease.

In this study, a storm event was set at one intensity to develop soil loss outputs, then an average long-term erosivity (R) was applied to determine K_{WEPP}. Undertaking further research over a range of storm event intensities to parameterise WEPP, as done by Silburn and Bosomworth (2023), to develop a range of K-factors against long-term erosivity factors would be of value. The seasonal variability of K, mentioned by several authors (Laflen 1982; Coote *et al.* 1988; Rosewell 1992; Auerswald *et al.* 2014; Benavidez *et al.* 2018), promotes the concept of running repeat simulations with varying rainfall intensities to produce soil loss observations and K-factors more representative of a long-term average. The K_{WEPP} developed here isolated the dominant interrill erosion process expected on hillslopes of low gradient based on observations from short slope-lengths. With longer slope-lengths and increased rainfall intensities, overland flow and flow power could increase despite the low gradient and introduce the potential for rilling to contribute to erosion. Investigating this process as a subsurface contribution to end-of-catchment would be useful.

While improvements of the K-factor will refine predicted soil loss from hillslopes when applied in catchment sediment modelling, these modifications themselves do not influence observed sediment loads at end-of-catchment. What they do facilitate is an improved understanding of the contribution of the hillslope sediment budget to the GBR lagoon and improved estimation of the spatial distribution of hillslope soil erosion, allowing for focus on either hillslope or other contributing sediment sources at the catchment scale. Further studies, similar to those presented here, will assist in obtaining a wider range of K-factors to capture variances from different soils and rainfall conditions, including the effect of antecedent rain. Where more field-measured K-factors are available for Queensland grazing lands, a new nomograph equation could be derived to better estimate erodibility for GBR catchment soils.

Conclusion

Soil susceptibility to erosion is complex and related to soil properties and influenced both by surface condition and seasonal fluctuations in rainfall intensity. Soil erodibility is often evaluated with nomograph equations based on plot-scale datasets from the USA, yet the applicability of the nomograph in Australia is limited, particularly on highly aggregated soils. Predicting long-term soil loss with the RUSLE and soil erodibility via calibrated WEPP and *in situ* field-measurements has not been widely attempted in Australia. Conducting rainfall simulation experiments to measure runoff and soil loss to derive an estimate of the long-term soil loss with WEPP could expedite the development of local soil erodibility estimates for hillslope erosion prediction in grazing lands, which are fundamental to modelling sediment budgets for the GBR catchment. Estimating the mean soil loss for the RUSLE using a WEPP-derived annual soil loss calibrated from plot-scale observations under rainfall simulation offers a cost-effective alternative to derive the K for RUSLE, provided that there are enough observed runoff and soil loss data to suitably calibrate the WEPP model. Erodibility factors modelled using WEPP soil loss data for the soils investigated in this study provided values in the low to moderate erodibility range, appropriate when considering the consolidation of surface soils. The K-factors derived here were up to 63% lower than those predicted with the nomograph and modified nomograph equations for most soils. This was attributed to the higher consolidation of the surface of grazing soils and detachment-limited erosion processes. This research provides support for the approach taken here as a viable alternative to the erodibility nomograph, especially for consolidated soils such as those found in grazing land across the GBR catchment.

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