Environmental Modelling and Software 147 (2022) 105228

Contents lists available at ScienceDirect

Environmental Modelling and Software

journal homepage: www.elsevier.com/locate/envsoft

Modelling soil loss from surface erosion at high-resolution to better understand sources and drivers across land uses and catchments; a national-scale assessment of Aotearoa, New Zealand

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ARTICLE INFO

Keywords:
Soil degradation
Surface erosion
Water quality
Grazing
Agriculture
RUSLE

ABSTRACT

Soil erosion is a significant challenge for agricultural regions, with cascading impacts to waterways, land productivity, soil carbon, and ecological health. We provide the first national-scale soil erosion model that incorporates the impacts of grazing on ground cover (C_{gr}) and soil erodibility (K_{tr}) into the RUSLE framework. Surface erosion rates for winter-forage paddocks (11 t ha^{-1} y^{-1}) were substantially higher than pastoral grasslands (0.83 t ha^{-1} y^{-1}), woody grasslands (0.098 t ha^{-1} y^{-1}), forests (0.103 t ha^{-1} y^{-1}) and natural soil production rates (≤1–2 t ha^{-1} y^{-1}). Validation with empirical measurements from sediment traps, sediment cores, and chemical fingerprinting demonstrated strong linear regressions ($r^2 = 0.86$). Terrain impacted soil erosion directly through slope steepness and flow convergence and indirectly through strong orographic effects on rainfall erosivity ($r^2 = 0.39$–0.83). Annual surface erosion across Aotearoa New Zealand could reach 16.5-29.2 Mt y^{-1}, representing ~$20M annually and up to 24–31% of sediment yield for two catchments.

1. Introduction

1.1. Background

Accelerated rates of soil degradation and erosion resulting from agricultural and pastoral land activities have been observed globally (Borrelli et al., 2017; FAO & ITPS, 2015; Smetanová et al., 2020) and are increasingly recognized as a threat to food production systems (Pimentel and Burgess, 2013), landscape stability (Trimble and Mendel, 1995), water quality (McCulloch et al., 2003), and ecosystem functioning (Larned et al., 2020). Natural rates of soil production and erosion under native vegetation are generally far less than soil loss rates from agricultural and pastoral landscapes by multiple-orders of magnitude (Hancock et al., 2020; Montgomery, 2007). Recent reviews have identified spatially and temporally explicit soil erosion models as one of the greatest opportunities for meeting soil conservation and Sustainable Development Goals (Lefèvre et al., 2020; Smetanová et al., 2020). To date, such models have not incorporated the effects of grazing on surficial erosion despite the well-documented impacts of increasing livestock grazing pressures on soil damage, accelerated soil and nutrient losses, and impaired water quality (Drewry et al., 2008; Greenwood and McKenzie, 2001; Hancock et al., 2020; Houlbrooke et al., 2009; Larned et al., 2020; McDowell et al., 2003; Merten and Minella, 2013; Monaghan et al., 2017).

In order to account for the impacts of livestock treading and grazing on surficial erosion, we apply a novel grazing model that captures the respective impacts of grazing and stock treading on soil physical properties and ground cover (Donovan and Monaghan, 2021) that integrates seamlessly with a seasonal and spatially explicit version the Revised Universal Soil Loss Equation (RUSLE). Incorporating grazing impacts on soil erosion will improve understanding of where landscapes are most and least susceptible to soil loss and degradation. In doing so, proactive decisions can help to minimize overlap between intensive landuse pressures and erosion-prone lands, rather than implementing costly reactive strategies.

1.2. Revised Universal Soil Loss Equation

RUSLE and its predecessor, USLE, predict mean annual soil loss from surface erosion based on a set of equations derived from empirical measurements of soil losses from agricultural plots (Renard et al., 1997). RUSLE encapsulates seasonal rainfall erosivity (R), slope length (L) and steepness (S), soil erodibility (K), ground cover and management factors (C); each of which is considered an important influence on soil loss

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https://doi.org/10.1016/j.envsoft.2021.105228
Received 4 February 2021; Received in revised form 29 July 2021; Accepted 8 October 2021
Available online 28 October 2021
1364-8152/

(Selby, 1993). Recent modelling developments have enabled additional precision through calculating seasonally-variable grazing-adjusted ground cover (C_{gr}) and treaded soil erodibility (K_{tr}) for pastoral lands (Donovan and Monaghan, 2021). Surface erosion models based on RUSLE are widely accepted and increasingly used for applications spanning field, catchment, national, and even global scales (Borrelli et al., 2017).

The components of the model include both inherent land characteristics including soil properties, slope length, and slope steepness, as well as allogenic (external) factors such as rainfall erosivity, ground cover and/or crop cover type, and some land management practices. Among the factors included in RUSLE, rainfall erosivity (R) is often most closely correlated with temporal trends in soil loss (Hedding et al., 2020), owing to the fact that sediments are rarely detached without sufficient rainfall. The impact of rainfall's kinetic energy on soil erosion is captured in the R-factor. Both rainfall volume and intensity should be included in calculating R due to the importance of short but highly erosive storm events (Nearing et al., 2005). We improve upon the calculation of R for New Zealand by using monthly rainfall data (NIWA, 2012) as input to seasonal and regionally variable linear and power functions (Klik et al., 2015) that were not previously available.

Slope steepness (S) seeks to capture the rate of change in soil loss with varying gradients, while slope length (L) accounts for the distance over which a slope gradient occurs. The equations for both S and L have been reformulated for use in geographic information systems (GIS) and are thoroughly summarized and compared (Bircher et al., 2019). Estimates of S and L for New Zealand are improved via the use of enhanced hydrological flow routing and empirical equations describing S and L found in literature. Soil erodibility is captured in the K-factor, which incorporates physical and chemical properties of soil including fractions of sand, silt, and clay, permeability, structural stability, and organic matter content. A recent global review of RUSLE formulations suggested that previous soil erodibility (K) factors for New Zealand (Dymond et al., 2010) did not adequately account for soil texture due to broad generalization (Benavidez et al., 2018). Herein, soil erodibility is enhanced by incorporating the soil structural vulnerability index (Hewitt and Shepherd, 1997) and additional soil physical and chemical properties, including particle size factions, surficial gravel and rock content, permeability, drainage classes, organic matter, and phosphate retention.

The cover and management factor (*C*-factor) is used to estimate the effect of canopy and ground cover, as well as land use management, in reducing surficial soil loss (Wischmeier and Smith, 1978). Known mechanisms through which cover impacts erosion include intercepting rainfall, slowing wind speed and transport, altering soil water content, providing root structure/cohesion, reducing surface water runoff, and adding soil carbon content. The effectiveness in reducing erosion varies spatially and seasonally with the height, density, and fraction of cover (Alexandridis et al., 2015; Basher et al., 2008; Benavides et al., 2009; Schmidt et al., 2018; Yang, 2014). C-factor estimates include added complexity when incorporating prior and current land practices such as tillage, crop residues, cover crops, and grazing that can each significantly alter soil exposure (Abdalla et al., 2018; Hoffman et al., 1983; Monaghan et al., 2017; Zhou et al., 2010). To date, formulations of the C-factor for New Zealand have only included 3 cover scenarios (i.e., bare ground, grass, and tree cover) that remain static throughout the year (Dymond et al., 2010), which does not capture the spatial or temporal variability across the many land covers of Aotearoa.

Despite the many strengths of RUSLE, it has limitations such that it does not capture gullying and/or shallow landslides, is not process based, does not capture feedbacks between model components, nor does it estimate delivery/deposition of eroded sediments. Being an empirical model that was developed in North America, RUSLE applications elsewhere should be validated and/or calibrated when estimating surface erosion. Previous reviews have covered these limitations in detail (Alewell et al., 2019; Benavidez et al., 2018) and further demonstrated that RUSLE outputs have similar ranges of uncertainty when compared with more complex process-based physical models such as WEPP and PESERA. Lastly, RUSLE has previously not accounted for the impact of grazing and treading on soil loss, despite significant impacts to ground cover and soil physical properties. Thus, we apply a nested grazing model (Donovan and Monaghan, 2021) to calculate treaded soil physical properties (RUSLE K_{tr}) and grazed ground cover (RUSLE C_{gr}). The grazing model uses empirical relationships between grazing/treading intensity (i.e., stock hoof pressure, grazing density, duration and history) and damage to soil physical properties (i.e., permeability and structure) and further account for susceptibility due to clay content and soil moisture (Donovan and Monaghan, 2021). Initial results demonstrated that this framework vastly improved estimates of soil loss from pastoral lands (Donovan and Monaghan, 2021), which is essential for accurately quantifying soil loss across New Zealand catchments, where pastoral lands occupy over 40% of land area (Fig. 1) (Ministry for the Environment and Stats NZ, 2019).

2. Material & methods

The surface erosion model is applied to Aotearoa, New Zealand (Fig. 1), with broad analyses encompassing 22 catchments (2400–21,960 km^2), followed by specific analysis of six prominent catchments with cultural and/or environmental importance: the Waikato, Rangitiaiki, Manawatu, Aparima, Clutha, and Motueka Rivers. The catchments were chosen to represent a range of environments, geologies, and characteristic land use distributions, specifically: annual and perennial croplands, grazed and ungrazed grasslands, natural and planted forests, and winter-forage paddocks. Descriptions of catchment soils, terrain, land use, and management can be found in (Donovan and Monaghan, 2021).

We used a 15-m digital elevation model (DEM) with seamless coverage of New Zealand, prepared and made publicly available by the National School of Survey at the University of Otago (Columbus et al., 2011). The DEM has vertical RMSE of $\pm$7.1 m and was interpolated from 20-m topographic contours using ANUDEM, a 2-D thin plate smoothing spline that optimizes for hydrologically-connected terrain while minimizing interpolation artefacts. Prior to any calculations of slope steepness (S) and length (L), spurious features were removed from the DEM to ensure proper flow routing. The formulation used herein is based on upslope contributing area (Desmet and Govers, 1996) and has been used for applications spanning field, watershed, national and even global scales (Borrelli et al., 2017; Panagos et al., 2014; Schmidt et al., 2019; Zhang et al., 2017). LS-factor values were limited to pixels with slopes <50% (26.6°) and drainage areas above 13,500 km^2 (equivalent to 60 cells of flow accumulations), similar to previously used thresholds found in literature (Bircher et al., 2019). LS-factor values were further masked for rivers and open waterbodies using New Zealand's hydrologic geodatabase. For additional details on calculating L and S factors, see Appendix A, Eqs. A.1-A.4.

Mean monthly rainfall grids based on New Zealand rainfall records spanning 1981–2010 (NIWA, 2012; Tait et al., 2012) were resampled using nearest neighbor interpolation from 100-m to 15-m resolution to align with the 15-m DEM. The 30-year rainfall data thus reflect the central tendency of each month's long-term conditions. Monthly grids were summed to calculate seasonal rainfall totals for Spring (September–November), Summer (December–February), Autumn (March–May), and Winter (June–August). Subsequently, we used previously derived linear and power functions to calculate seasonal rainfall erosivity (R) (Klik et al., 2015), which captures the kinetic energy potential of rainfall that drives soil erosion by water. The linear and power function coefficients vary seasonally and spatially (e.g., Table 3 of Klik et al., 2015) across four distinct climatic zones of New Zealand (Fig. 2). Each was derived using mean monthly rainfall, maximum 30-min rainfall intensity, and rainfall erosivity at 632 weather stations, with r^2 of 0.82–0.98. Additional details on rainfall erosivity (R) calculations are provided in Appendix B and Klik et al. (2015). We explore the role of

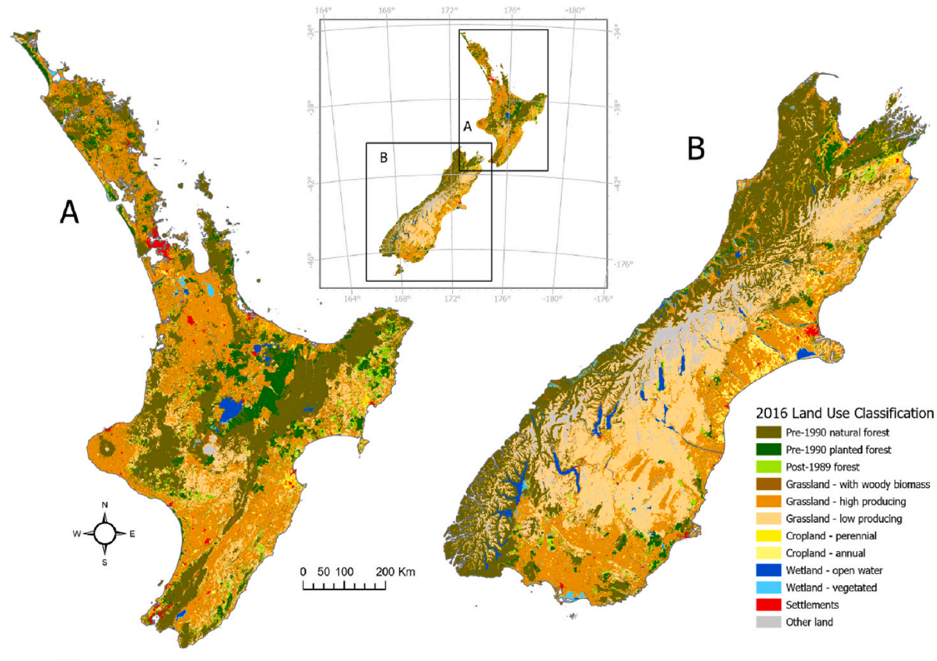

Fig. 1. Land cover and land use map for the North and South Islands of New Zealand. Land use data was obtained from the 2016 Land Use and Carbon Analysis System (LUCAS) geodatabase, with the exception of Winter forage paddocks (purple), which were derived from satellite imagery acquired in 2018 (Belliss et al., 2019).

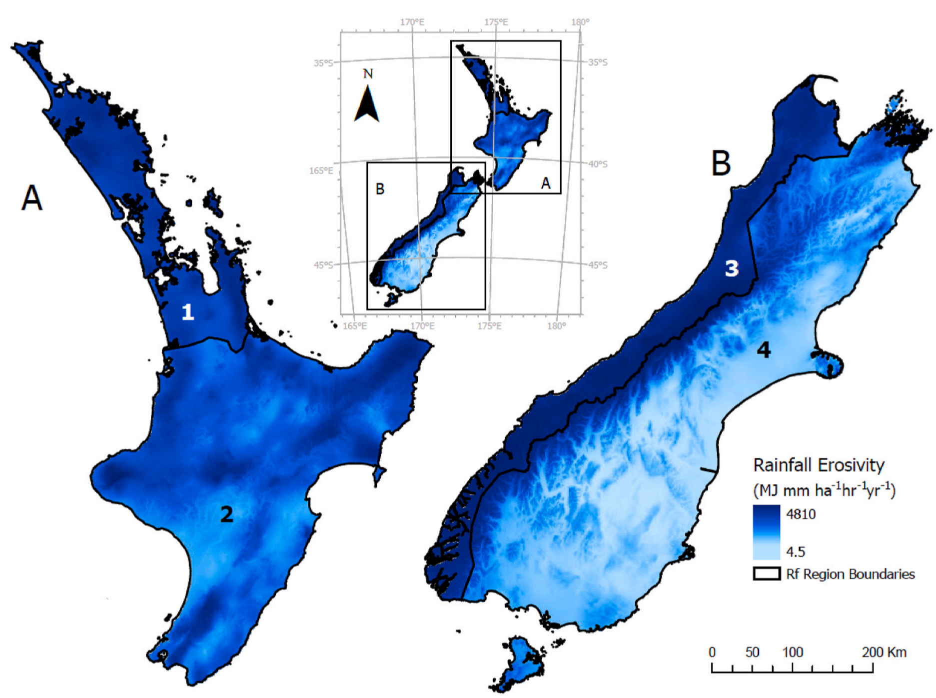

Fig. 2. Rainfall erosivity (R) map for the North (A) and South (B) Islands of Aotearoa, New Zealand. Rainfall erosivity gradient and values reflect winter conditions. Black boundaries and numbers (1–4) indicate unique climate and rainfall subregions as determined in Klik et al. (2015). For each region, a unique formulation to calculate rainfall erosivity (R) was applied.

elevation on rainfall intensity by evaluating the significance of regressions between elevation (meters above sea level) and seasonal rainfall intensity ($\log_{10}$, to normalize data) for four regions with distinct rainfall erosivity patterns (Klik et al., 2015). We compared a series of regressions that included the additive effects of location (X- and Y-coordinates) to confirm the most parsimonious model using AIC and BIC criteria, where lower scores indicate more optimal model fit and complexity.

The Land Use and Carbon Analysis System (LUCAS) 2016 geodatabase was used to classify land use and land cover types across New Zealand (Fig. 1). LUCAS is derived from 10-m Sentinel 2 satellite imagery, further refined using data fusion with other New Zealand landuse geodatabases (Newsome et al., 2018). Herein, LUCAS was also used to omit land areas not suited for the RUSLE model or those not susceptible to surface erosion from overland flow, such as open water bodies, wetlands, urban areas/settlements, bedrock outcrops, sand dunes, beaches, coastal cliffs, mines or quarries, and permanent ice or glaciers. We assume the spatial error in landuse boundary delineations are negligible across national scales (Donovan et al., 2019). Baseline values of the cover and management factor (C) for each crop and cover class were derived from a comprehensive review of relevant measurements found in previous studies (Basher et al., 2016; Gabriels, 2003; Panagos et al., 2015b; Puente et al., 2011; Vatandaşlar and Yavuz, 2017), with seasonal adjustments to account for typical growth and senescence. For pastoral land covers, a nested sub-model was applied to account for differences in the density and fraction of cover following grazing, denoted as $C_{gr\text{-}season}$. Cover and management factor (C & C_{gr}) values are summarized in Table 1, along with details and calculations within Appendix C (Eqs. C.5 & C.6), and Donovan and Monaghan (2021).

Soil characteristics used to calculate soil erodibility (K) factor were derived from the Fundamental Soil Layer (FSL), a free and publicly-available national soil map (Newsome et al., 2008) with physical, chemical, and mineralogical information. The maps were compiled at 1:63,360 (inch to mile) from 1500 soil profiles that were refined using local surveys, topographic maps, and aerial photos. Much of the methods, data attributes and categories are described in the LRIS Data Dictionary (Newsome et al., 2008) and were derived from either Webb and Wilson (1995) or Clayden and Webb, 1994. Inherent soil erodibility (K) was calculated using the standard derivation (Eq. (1); Renard et al., 1997), with additional adjustments for stoniness (K_{st}) found in (Panagos et al., 2014; Poesen and Ingelmo-Sanchez, 1992; Poesen et al., 1994).

$$K = \left[\frac{2.1\times10^{-4}M^{1.14}(12-OM)+3.25(s-2)+2.5(p-3)}{100}\right]*0.1317 \tag{1}$$

For pastoral lands, we calculated a seasonal grazing-adjusted soil erodibility (K_{tr}) using recent livestock treading model (Donovan and Monaghan, 2021) that captures the change in two subfactors-soil structure (s_{tr}) and permeability (p_{tr})- that are impacted by livestock treading (Eq. (2)). The model uses grazing information (livestock hoof pressure, stocking density, grazing duration, and grazing history), along with soil properties (structural vulnerability, soil moisture, soil texture). The Supporting Material (e.g., Section 1.4) describes the details of each soil attribute used to calculate inherent (K) and treaded soil erodibility (K_{tr}).

$$K_{tr} = \left[\frac{2.1\times10^{-4}M^{1.14}(12-OM)+3.25(s_{tr}-2)+2.5(p_{tr}-3)}{100}\right]*0.1317 \tag{2}$$

Table 1
Seasonally variable values of C_{crop} and Fr_{cover} (fraction of cover) for each land cover type found in the LUCAS geodatabase. In addition to seasonal values of C_{crop} and Fr_{cover}, we include a column for post-grazed cover values. An additional LUCID (83) was added for areas known to have winter grazing (Belliss et al., 2019).

Land Use Class	Subclass Description	Seasonal C_{crop} & (Fr_{cover}) Values						
		Grazed	September–November	December–February	March–May	June–August		
			C_{spring}	C_{summer}	C_{autumn}	C_{winter}		
Natural forest	Unknown Wilding trees; tall indigenous forests	–	0.002 (1.0)	0.0012 (1.0)	0.0012 (1.0)	0.003 (1.0)		
Planted forest, pre-1990	Unknown Pinus radiata Douglas fir Exotic species	–	0.004 (1.0)	0.003 (1.0)	0.003 (1.0)	0.005 (1.0)		
Planted forest, post-1989	Unknown Wilding trees; tall indigenous forests Pinus radiata Douglas fir Exotic species Regenerated natural species	–	0.005 (1.0)	0.004 (1.0)	0.004 (1.0)	0.007 (1.0)		
Grassland with woody biomass	Grassland with matagouri and sweet briar, broadleaved hardwood shrubland, manuka/kanuka shrubland, coastal and woody shrubland	–	0.01 (1.0)	0.009 (1.0)	0.008 (1.0)	0.01 (1.0)		
Grassland high producing; grassland with high quality pasture species	Grazing, dairy	Ungrazed	0.02 (0.95)	0.01 (0.99)	0.01 (0.95)	0.03 (0.90)		
		Grazed	0.04 (0.80)	0.03 (0.75)	0.03 (0.78)	0.05 (0.78)		
	Grazing, non-dairy	Ungrazed	0.02 (0.95)	0.01 (0.99)	0.01 (0.95)	0.03 (0.90)		
		Grazed	0.04 (0.90)	0.03 (0.85)	0.03 (0.88)	0.05 (0.87)		
	Ungrazed	–	0.02 (0.95)	0.01 (0.99)	0.01 (0.95)	0.03 (0.90)		
	Unknown	–						
Grassland low producing; low fertility grassland and tussock grasslands; mostly hill country	Grazing, dairy	Ungrazed	0.03 (0.95)	0.02 (0.99)	0.02 (0.95)	0.04 (0.90)		
		Grazed	0.05 (0.80)	0.04 (0.75)	0.04 (0.78)	0.06 (0.78)		
	Grazing, non-dairy	Ungrazed	0.03 (0.95)	0.02 (0.99)	0.02 (0.95)	0.04 (0.90)		
		Grazed	0.05 (0.90)	0.04 (0.85)	0.04 (0.88)	0.06 (0.87)		
	Ungrazed	–	0.03 (0.95)	0.02 (0.99)	0.02 (0.95)	0.04 (0.90)		
	Unknown	–						
Cropland, perennial	Orchards or vineyards	–	0.3 (0.85)	0.28 (0.90)	0.33 (0.90)	0.35 (0.60)		
Cropland, annual	Annual crops and cultivated bare ground	–	0.3 (0.75)	0.28 (0.90)	0.33 (0.90)	0.35 (0.40)		
All open water, i.e., lakes, rivers, dams, reservoirs, estuaries	Unknown Natural Human induced	– – –	0.0 (1.0)					
Wetland or non-forest woody vegetation in a wetland context	Unknown Peat mine	– –						
Settlements, built up areas or impervious surfaces, roads	–	–						
Other	Rock outcrops, dunes, beaches, quarries, permanent snow, or glaciers	–					82	0
Winter-forage crop, Brassica	Grazed, intensive, Brassica- fodder beet, swede, or kale	Ungrazed	0.05 (0.80)	0.04 (0.75)	0.04 (0.78)	0.06 (0.78)	83	505
		Grazed	0.4 (0.0)	0.36 (0.3)	0.33 (0.6)	0.3 (0.0)		

For each season, soil loss from surface erosion (E_s, Eq. (3)) was calculated as the product of all factors (R_{season}, L, S, K or $K_{tr\text{-}season}$, and C_{season} or $C_{gr\text{-}season}$), which were then summed to produce grids of annual erosion (E_{yr}, Eq. (4)).

$$E_{season} = R_{season} * K_{tr-season} * L * S * C_{gr-season} \quad (3)$$

$$E_{yr} = E_{Sp} + E_{Su} + E_{Au} + E_{Wi} \quad (4)$$

Where E_{Sp}, E_{Su}, E_{Au}, and E_{Wi}, are soil losses for the Spring, Summer, Autumn, and Winter. We omitted non-erosive areas and those considered unsuitable for RUSLE modelling, including urban developments, glaciers, bedrock exposures, mountainous environments, mines, quarries, wetlands, and open water. For additional details on the data, processing algorithms, and software used, see Appendices A-E.

For evaluating accuracy and uncertainty of the soil loss estimates, the annual soil loss grid was clipped and summed for specific paddocks, farms, land use classes, and/or catchments with published measurements of soil loss. The studies used for assessing uncertainty included all comparable, known, research on soil losses measured using in-field sediment traps, flumes, storm and baseflow sampling, estuary and lake sediment cores, and Caesium-137 areal distributions (Fig. 8, Table 3). Finally, we use recent economic analyses that valued the cost of surficial soil loss at \$1.2 NZD per tonne (Soliman and Walsh, 2020) to calculate the total cost associated with surficial soil losses across New Zealand. Specifically, the product of total soil losses for New Zealand (tonnes) and average cost rate (\$1.2 NZD per t yr^{-1}) yielded the cost associated with surficial soil loss. The cost estimates only account for the value of soil lost, not costs associated with environmental remediation or loss of farm productivity.

3. Results & discussion

3.1. Rainfall erosivity (R)

Across New Zealand, annual rainfall erosivity varied from <100 to >20,000 MJ mm ha^{-1} hr^{-1} yr^{-1} (Fig. 2, Table 2) with mean values of 2481 and 2952 for the North and South Islands, respectively. The mean annual values for each Island lie at the high end of the range for Oceanic climates ('*Cfb*' in the Köppen climate classification) found in previous global rainfall analyses (Panagos et al., 2017; e.g., Fig. 3). Extremely high values (>10,000) exist within small portions of New Zealand's Southern Alps, comparable to some of the highest values found worldwide in regions such as the Amazon (da Silva, 2004; Panagos et al., 2017). The spatial distribution, seasonal variation, and range of R-factor values align with previous values modelled for New Zealand (Klik et al., 2015), confirming the precision and seasonal distribution of rainfall erosivity results despite using distinct input datasets. Seasonally, both islands experience maximum rainfall erosivity in Summer, while the lowest quartile of values occurred in Spring. Seasonal variability in mean R-factor values were slightly different; the highest mean rainfall erosivity for the North Island occurred during Autumn, while average erosivity for the South Island peaked during Summer. In other words, maximum potential erosive power of rainfall does not coincide with the period during which the majority of land area is experiencing elevated erosive forces from rainfall.

Terrain is known to have an important role in rainfall quantity at local scales via orographic effects (Hutchinson, 1968), however, this effect was thought to be negligible at sufficiently broad scales (Tait et al., 2006). Within each region (Fig. 2) elevation explained 14–50% of the variability in log-rainfall erosivity based on linear regressions, which was improved further (39–86%) when incorporating latitude and longitude (Supplementary Material, Appendix B, Figs. B1-B5). This is the first analysis and results to show such a pronounced impact of terrain on rainfall erosivity over broad spatial scales across New Zealand.

3.2. Terrain; slope length and steepness factors (LS)

The methods used herein are the first national scale map of the LS-factor for Aotearoa, New Zealand, providing a high-resolution map of how terrain (slope length and steepness) impact risk to surficial erosion. Respectively, the North and South Islands had mean LS-factor values of 1.62 and 1.29 (σ = 1.83 and 1.91), which are comparable to the central tendency (μ = 1.63) of the European Union (Panagos et al., 2015a). While the vast majority of land area across New Zealand has low LS-factor values (Fig. 3), the Southern Alps give rise to more land with intermediate to high LS-factor values (Fig. 3B) compared to the North Island. This is evident in a 25% higher mean LS-value for the South Island compared to the North Island, which is equivalent to a 25% increase in average susceptibility to surface erosion arising from topographic differences alone.

3.3. Soil erodibility (K & K_{tr})

Soil erodibility values (K) are directly proportional to actual erodibility, so higher K-values indicate soils are more susceptible to being eroded, and vice versa. Soil erodibility exhibited moderate spatial variability, with bimodal distributions for both islands (Fig. 4). Mean and median K_{tr}-values were 0.029 (σ = 0.018) for the South Island, which were similar, but slightly higher than the North Island (0.020, σ = 0.014). The mean values are slightly lower than that of the European Union (μ = 0.032), which may reflect the significant impact of the Loess belt across EU, which is generally associated with higher erodibility (Panagos et al., 2014). New Zealand soil erodibility was generally inversely related to permeability (Fig. 5B) and directly related to dominant particle size (Fig. 5C), indicating agreement with expected trends of soil erodibility. K_{tr}-values were previously validated by comparing the change in soil erodibility with field-measurements of pre- and post-grazing changes in soil macroporosity for a range of grazing pressures spanning pastures and forage crops (Donovan and Monaghan, 2021). The inclusion of stoniness reduced mean soil erodibility by 4.5%, demonstrating the importance of including the stoniness effect into soil erodibility calculations. Across grazed areas of New Zealand, the effect of including livestock treading increased soil erodibility (K_{tr}) by 3.3–9%, with the most pronounced effects occurred in winter due to the impacts of intensive forage crop grazing practices.

Table 2
Summary statistics of rainfall erosivity values for each season and Island of Aotearoa, New Zealand.

	Rainfall erosivity (R_f) Values									
	South Island					North Island				
	Min	Max	Mean	Std	Fraction	Min	Max	Mean	Std	Fraction
Spring (Sep–Nov)	16	5710	641.7	878.1	22%	105	2847	474.3	214.9	19%
Summer (Dec–Feb)	63	8857	917.0	1299.7	31%	136	3402	655.3	255.1	26%
Autumn (Mar–May)	14	7466	854.9	1123.9	29%	184	3219	777.24	299.8	31%
Winter (Jun–Aug)	4.5	4876	547.2	672.9	18%	144	3759	587.1	271.5	24%
Year (Annual)	138	26,382	2952.3	3927.0	100%	644	13,227	2481.5	1012.3	100%

Table 3
Summary of uncertainty analysis comparisons, including locations, measurement method(s) for soil losses, spatial and temporal scales of measurement, comparable modelled soil loss for the same extent, and the source of cited/measured soil loss. Numerical superscripts next to each source align with those found in Fig. 8.'

Dominant landuse/cover	Location	Method	Spatial scale/area	Timescale	Cited soil loss (t ha^{-1} yr^{-1})	Modelled soil loss (t ha^{-1} yr^{-1})	Source
Grassland, grazed	Waikato, Manawatu, Otago, Bay of Plenty, Southland	Varied	<1–25 km^2	Varied	0.022–4.40	11.6–251.7	[1]McDowell and Wilcock (2008)
Grassland, grazed	Multiple	Varied	Catchment	Varied	0.60–2.0		[1]Elliott and Carlson (2004); [1]Vant (2001)
Grassland, grazed	Otago	Sediment traps	<1 km^2	2 yr	0.70–1.70	107.0	[2]Cournane et al. (2011)
Grassland, grazed	Tamingimingi catchment	Sediment traps	8 km^2	10 yr	0.60–1.40	135.0	[3]Fahey et al. (2003); [3] Fahey and Marden (2006)
Grassland, grazed	Mangaotama catchment	Sediment traps	3 km^2	12 yr	0.55–1.57	43.3	[4]Hughes et al. (2012)
Pastoral uplands, grazed	Tutira catchment	Lake core sedimentation	29.6 km^2	38 yr (1963–2001)	5.7	5.4	[6]Page et al. (2004)
Pastoral hill country, grazed	Manawatu catchment	Sediment trench & H-flume	<1 km^2	19–53 months	1.10–2.74	1.2	[5]Lambert et al. (1985)
Winter-forage crop, grazed	Telford farm	Sediment traps	<4 ha	3 yr	4	4.01	[7]Monaghan et al. (2017)
Winter-forage crop, grazed	Taupaka farm	Sediment traps	<0.5 ha	1 yr	0.17–0.829	0.83	[8]Burkitt et al. (2017); [8] Fransen et al. (2017)
Winter-forage crop, grazed	Taupaka farm	Sediment traps	<0.5 ha	1 yr	0.43–4.61	0.997	[9]Burkitt et al. (2017); [9] Fransen et al. (2017)
Tussock grasslands	Otago	Sediment weir	2.1–3.3	4 yr	0.01–0.06	0.019–0.105	[10]O'Loughlin et al. (1984)
Mixed pasture and forest	Mahurangi catchment/ estuary	Estuary core deposition	121 km^2	19 yr	1.25 (0.29–3.26)	0.39 (0.31–1.29)	[11]Oldman et al. (2009)
Forested	Golden Bay	Sediment traps	<1 km^2	2 yr	0.13	0.11	[12]O'Loughlin et al. (1978)
Cropland	Pukekohe	^{137}Cs - Caesium-137 activity		3 yr	0.70–3.0	1.28	[13]Basher and Ross (2002)

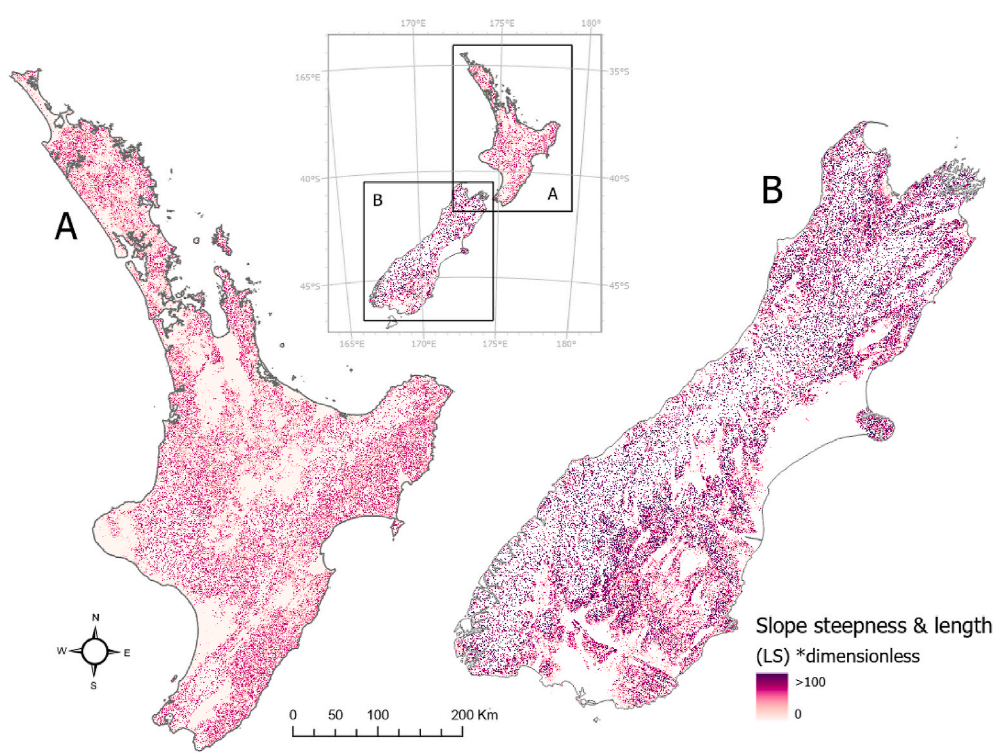

Fig. 3. Slope length and steepness for the North (A) and South (B) Islands of Aotearoa, New Zealand. Note that LS-factor values were filtered for areas with slopes >50°, flow accumulation >13,500 m^2, and for known waterways and waterbodies. Thus, areas that may be expected to exhibit the highest LS-factor values, such as the Southern Alps, appear as having low (0) LS-factor values relative to areas with significantly less relief.

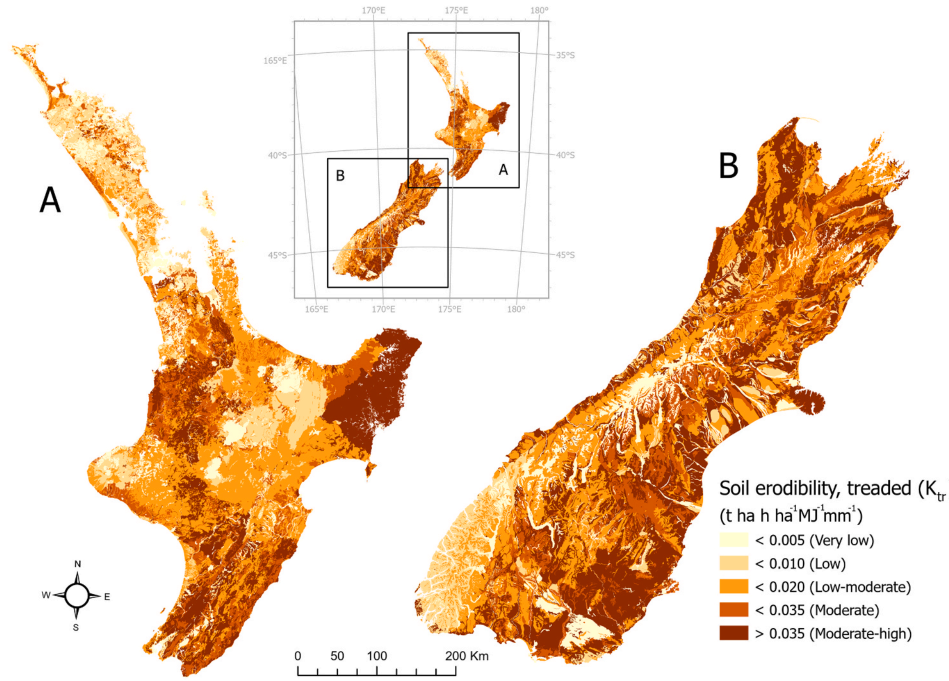

Fig. 4. Soil erodibility ($K_{st\text{-}tr}$) map of Aotearoa, New Zealand. In addition to the typical variables included in calculating soil erodibility (clay, silt, very fine sand, organic matter, structure, and permeability) the values calculated herein also account for seasonally-averaged soil moisture, drainage, phosphate retention, surface stoniness, and livestock treading effects on soil properties, where applicable.

3.4. Cover & management factor

The range of cover factors (C_f) across New Zealand varied from 0 to 1, with similar variability for both North and South Islands (Fig. 6). C-factor values are inversely proportional to the effectiveness in reducing erosion; thus, C-factor values of 0 reflect 100% reductions in surface erosion while 1 reflects no reduction in surface erosion. Differences across seasons were minor apart from croplands and winter-forage paddocks, which exhibit strong variation due to harvesting, senescence, and/or grazing. Significant variability existed across land use and cover classes, reflecting the effectiveness of each vegetation in mitigating surface erosion. Forests were consistently the most effective in reducing surface erosion, followed by grasslands with woody biomass, ungrazed grasslands, grazed grasslands, perennial and annual croplands, and winter forage crop paddocks (Fig. 7).

Prior to grazing, low and high-producing grasslands exhibit negligible differences in C_f values (blue boxplots, Fig. 7). However, after grazing, mean C_{gr} values were 53% greater for high-producing grasslands (orange boxplots, Fig. 7), equivalent to doubling in susceptibility to surface erosion. Further investigation revealed the increased susceptibility for high-producing grasslands reflected the predominance of grazing by dairy cattle, which generally leave ~10% less residual ground cover (75–80% cover) compared to grazing by sheep and beef cattle (85–90% cover) (Elliott and Carlson, 2004; Pande et al., 2000). Based on empirical relationships between ground cover and soil loss (Silburn et al., 2011), this modest (10–15%) gap in ground cover is equivalent to a 53% difference in mean annual soil loss. These results demonstrate how retaining ground cover residuals following grazing can be used as a simple means to enhance soil retention in grazed lands.

3.5. Uncertainty and validation

In order to provide a thorough and transparent view of the uncertainty associated with RUSLE surface erosion outputs, we expand upon uncertainty analyses presented in Donovan and Monaghan (2021). Herein, we go beyond pastoral grasslands and winter-forage crop paddocks to incorporate mixed landuse catchments (Oldman et al., 2009), forested lands (O'Loughlin et al., 1978), croplands (Basher and Ross, 2002), and natural tussock grasslands (O'Loughlin et al., 1984). The studies used for assessing uncertainty include all comparable measurements of soil losses that are dominated by surficial erosion processes (rill and interrill erosion), rather than gullying and/or shallow landslides. We compare with studies using a diverse set of measurement techniques, spanning in-field sediment traps, flumes, storm and baseflow sampling, estuary and lake sediment cores, and Caesium-137 areal distributions (Fig. 8, Table 3). Comparisons between measured and modelled soil losses indicate that modelled rates fall within the range measured at each field location found in literature (Fig. 8A). This is supported further by a strong linear regression ($r^2 = 0.86$) between measured and modelled rates (Fig. 8B). Additional details on the studies used for validation and uncertainty analyses are provided in Appendix E. Ongoing work is underway to validate, improve upon, and assess uncertainty in calculated soil losses at farm and catchment scales.

From these comparisons, we can suggest that the soil losses modelled via the grazing-adjusted RUSLE framework adequately capture the magnitude of soil losses expected under average rainfall conditions for the range of land uses considered. The broad-scale comparisons herein and in Donovan and Monaghan (2021) demonstrate significant alignment between modelled and measured long-term surficial soil losses from surface erosion where such losses are transported. This is further supported by favorable alignment between spatially-explicit comparisons between modelled and measured soil losses. RUSLE-based modelled soil losses represent total potential soil losses from surficial erosion processes, and thus, we do not suggest they represent sediment transport or the total soil losses entering waterways of New Zealand. Future work should assess the fate of surface erosion using sediment transport modelling alongside bank erosion measurements from aerial imagery to gain a more complete picture of catchment sediment sources.

Six subcatchments within Manawatu Catchment-which have been the focus of numerous monitoring and modelling studies-are used to understand how surficial erosion compares to total catchment sediment yields (Fig. 9). This comparison provided the first estimate of potential

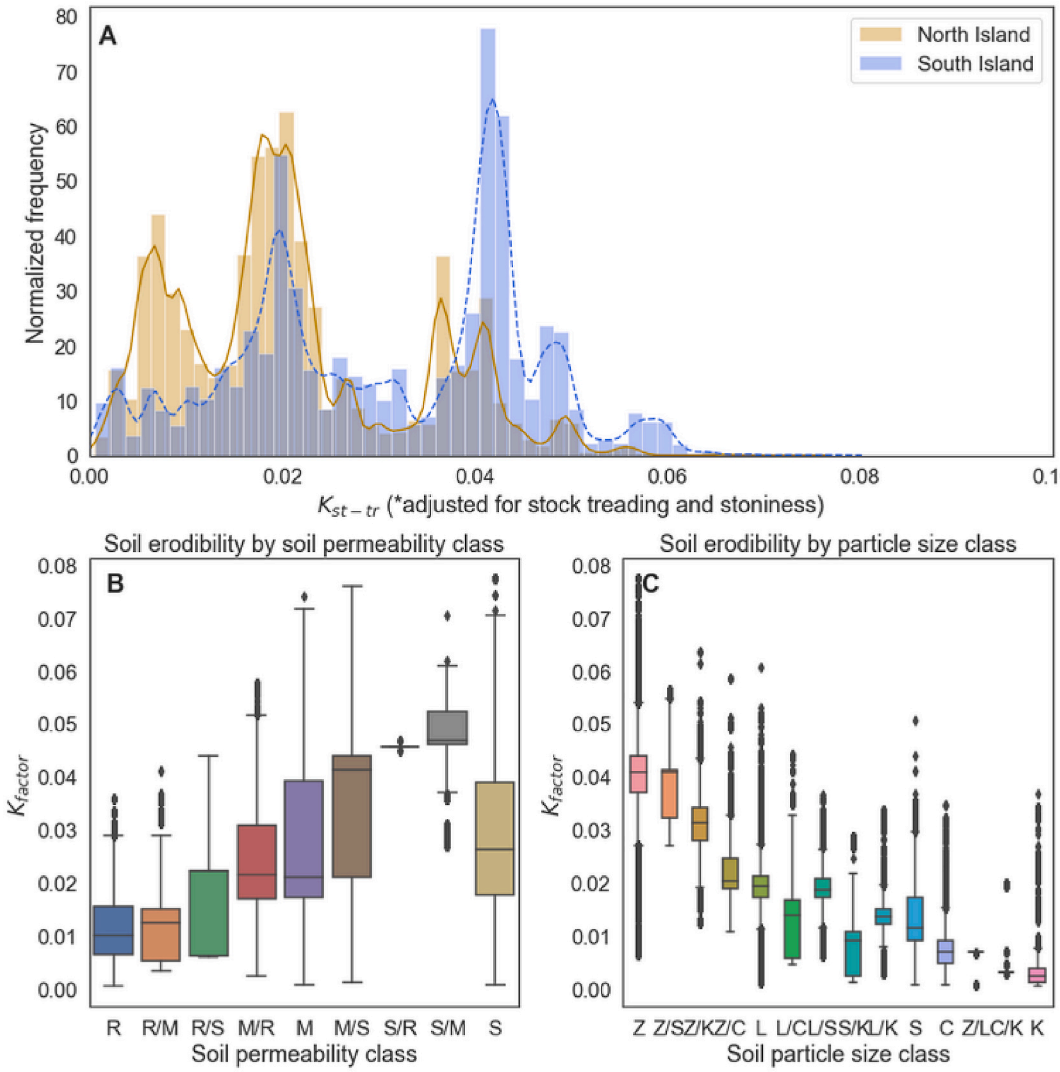

Fig. 5. Overview of soil erodibility ($K_{st\text{-}tr}$) across Aotearoa, New Zealand. (A) Normalized frequency distributions of soil erodibility for both the North (yellow) and South (blue) Islands. (B) Soil erodibility distinguished by soil permeability classes, as defined in the Fundamental Soils Layer (FSL). (C) Soil erodibility values for each dominant particle size class (Z = silty, S = sandy, C = clayey, L = loamy, K = skeletal). These results confirm that soil erodibility generally follows expected trends that match gradients in permeability and particle size.

contributions from surface erosion over timescales sufficiently long to assume 100% sediment delivery. Over annual timescales, this approach cannot ascertain how much modelled surface erosion is contributing to sediment yield due to temporal lags in sediment delivery of rivers impacted by land use change (Donovan et al., 2021; Donovan and Belmont, 2019), and because RUSLE does not account for sediment transport and deposition. Thus, the comparisons likely reflect the high end of potential contributions to annual sediment yields found in literature (Vale et al., 2016).

Modelled surface erosion across all six subcatchments represented 30% of the total annual sediment yield measured at each outlet, which varied from 23–48% of downstream sediment yields in five of the six subcatchments (Vale et al., 2016). For the sixth (Mangatainoka subcatchment), modelled soil losses were 73% greater than downstream yields, which is similar to previous modelling efforts with SedNetNZ (Dymond et al., 2014; Vale et al., 2016). The consistency of this overestimate suggests that either significant deposition occurs, measurements underestimate sediment contributions (Vale et al., 2016), or land conditions exist within the catchment that are not captured within the models or input datasets.

Mapping potential hotspots of soil loss across the catchment (Fig. 9) illustrates how sparse vegetation cover along steep uplands in the northwest portion of the Pohangina subcatchment may be driving elevated surface erosion rates. An abrupt shift along the boundary of the Pohangina and Upper Manawatu catchments with dense vegetation cover appear much less susceptible, evident in the abrupt spatial transition from orange/red to green hues. We confirmed this shift is not explained by changes in slope, soils, and/or rainfall erosivity. Southwest portions of the Manawatu catchment also exhibit low rates of surface erosion and are thus unlikely to contribute significant sediment to the river. This excludes streambank erosion, which is often a significant source of sediment remobilized into downstream estuaries (Donovan et al., 2015, 2016).

3.6. *Soil loss contributions across catchments and land uses*

Modelled soil losses for all lands subject to surface erosion processes across New Zealand (Fig. 9) have the potential to reach 16.5-29.2 Mt y^{-1} (Fig. 10, Table 4). Over timescales sufficiently broad to encompass transport to rivers, these could account for 15% of the 192 Mt y^{-1} of sediments estimated to reach waterways and oceans, annually (Ministry for the Environment & Stats NZ, 2018). While eroded sediments may not

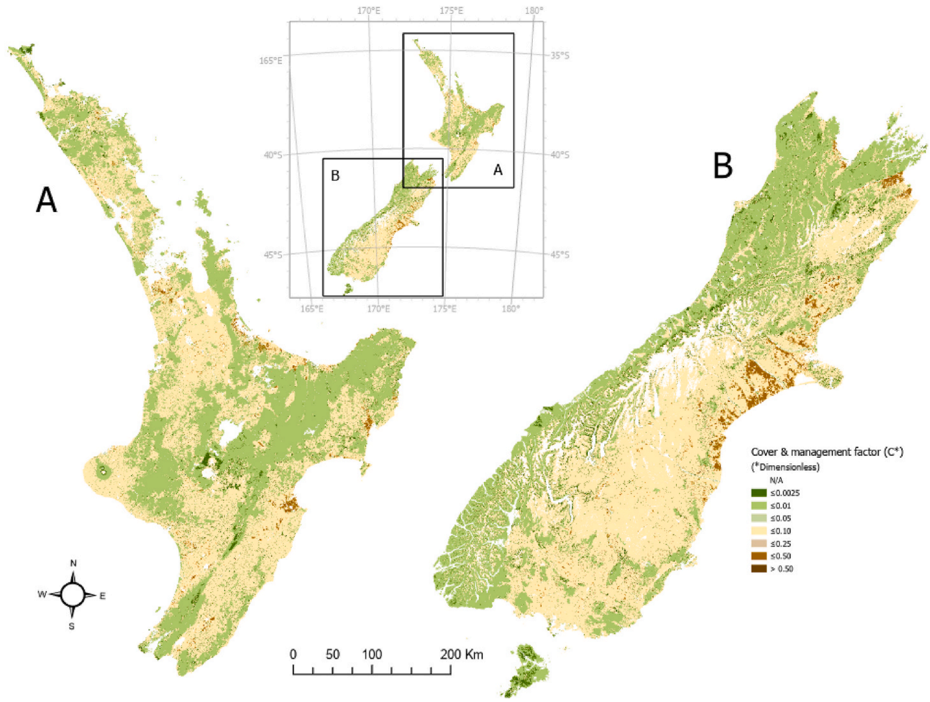

Fig. 6. Cover and management factor for the (A) North and (B) South Islands of Aotearoa, New Zealand. Values illustrated are for Winter. C-factor values for each vegetation type vary with seasonal growth patterns, as described in Table 2. White areas are those without cover factors, which include lakes, urban areas, bedrock, and glaciers or permanent snow.

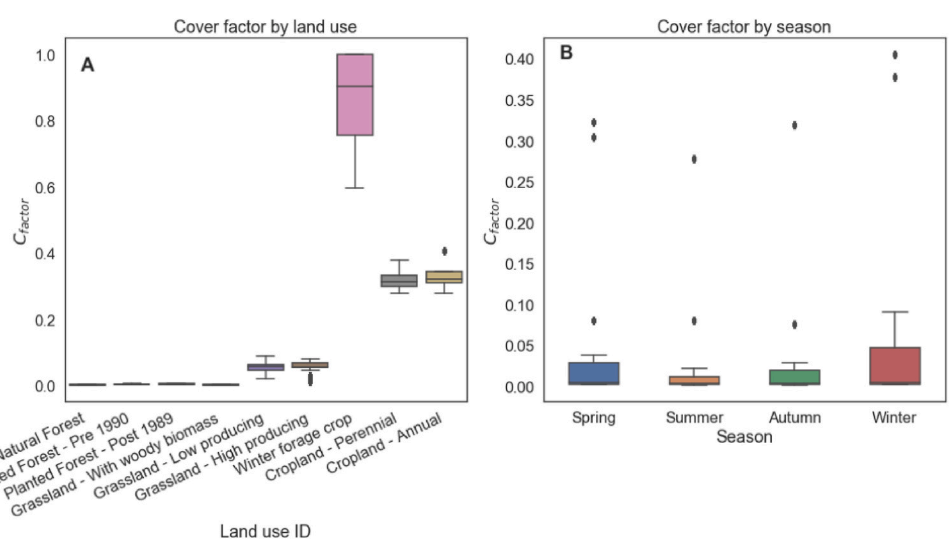

Fig. 7. Distinct cover factor value distributions for each land use/cover class included in the final surface erosion model. Lower C_{factor} values are indicative of cover that is more effective at reducing erosion. Differences within landuse/cover types reflects seasonal growth/senescence, grazing intensity or management, and/or harvesting.

reach waterways over sub-decadal timescales due to intermediary deposition, such soil losses nevertheless are a loss of significant social, spiritual and economic value to communities of Aotearoa, New Zealand, especially Māori (Asher and Naulls, 1987; Harmsworth, 2020; Hutchings et al., 2018). Previous economic evaluations valued New Zealand soils at an average of \$1.2 NZD per t y^{-1} (Soliman and Walsh, 2020). Applying this value to the soil loss estimates herein indicates that surficial soil losses represent a loss of ~ \$20-40M annually, excluding the environmental costs of remediation and losses in agricultural productivity.

For the 22 catchments considered, the average soil loss via surface erosion from pastoral grasslands was 0.84 t ha^{-1} y^{-1} and ranged from 0.02 to 2.5 t ha^{-1} y^{-1} (Fig. 11), which is lower than average soil losses for the European Union (2.02 t ha^{-1} y^{-1}) and globally (1.70 t ha^{-1} y^{-1}) (Doetterl et al., 2012; Panagos et al., 2015c). Within the six catchments examined further, these rates accounted for 34–92% of the total soil loss from surface erosion (Fig. 11), while the fraction of pastoral grassland landcover occupied 15–76% of the catchments' erodible area. On the other hand, winter forage crop paddocks on slopes greater than 7° were <1% of the catchment areas, but accounted for up to 12% of the catchments' total surface erosion (Fig. 12). Put another way, grazed forage-crop paddocks contribute 7- to 120-fold more soil loss relative to proportion of land area they occupy. The average rates of soil loss (11 t ha^{-1} y^{-1}) from forage-crop paddocks were the highest of any land

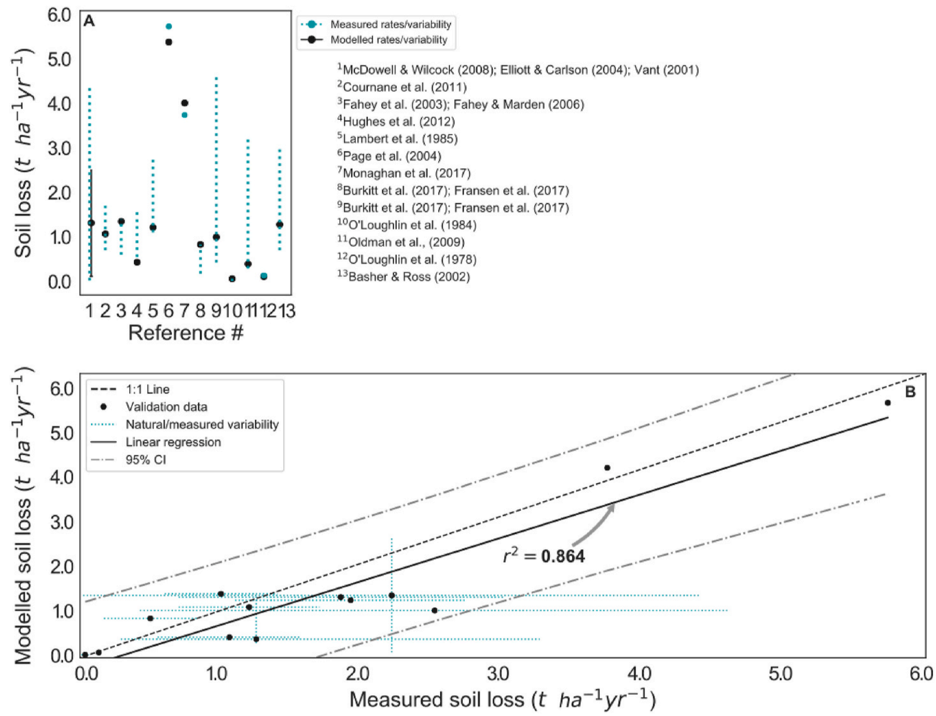

Fig. 8. Model validation data comparisons. (A) Visual comparisons of modelled (black, solid) and measured (green, dotted) rates and the natural/measured variability found at each location. (B) A strong linear regression ($r^2 = 0.864$) between measured and modelled values indicated strong model alignment with field-based measurements.

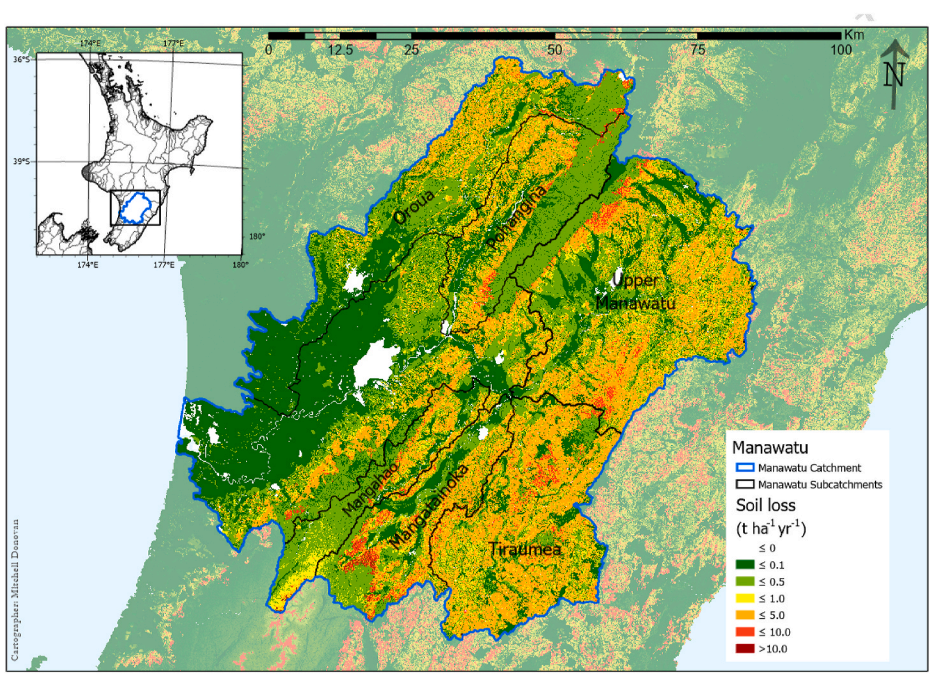

Fig. 9. Map of modelled soil loss via surface erosion for the Manawatu catchment and six subcatchments: Oroua, Pohangina, Upper Manawatu, Tiraumea, Mangatainoka, and Mangahao. White areas are those which do not experience surface erosion or cannot be modelled using the RUSLE framework, including open water bodies, wetlands, urban areas, glaciers, exposed bedrock, quarries, and beaches/dunes.

use/cover modelled, with considerable variability (3.90–24.52 t ha^{-1} y^{-1}). Previous analyses of soil losses from forage-crop paddocks demonstrated that such high rates reflect the combination of negligible ground cover following grazing, intensive treading on wet soils, and relatively steep terrain (Donovan and Monaghan, 2021).

Natural ungrazed grasslands had highly variable rates of soil loss ($\mu = 2.0$, range $= 0.02$–10 t ha^{-1} y^{-1}), and generally contributed higher proportions of surface erosion relative to their fractional land area. Additional investigation revealed that natural grasslands generally inherit high rates of surface erosion from steep slopes and highly erodible soils (Donovan and Monaghan, 2021), which was the same for natural grasslands across the European Union (Panagos et al., 2015c).

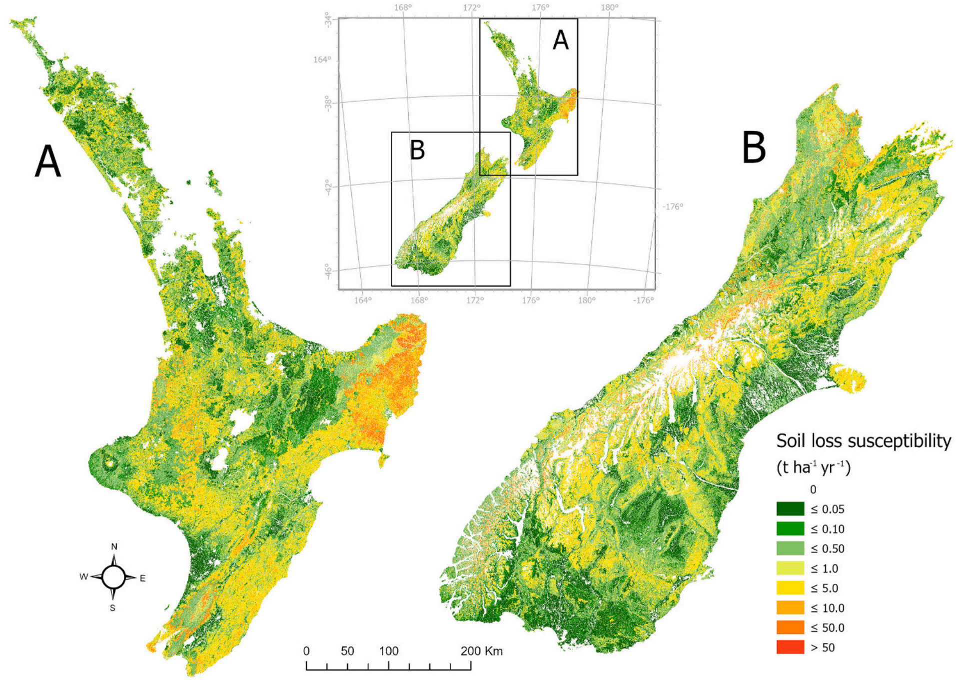

Fig. 10. National map of modelled soil loss from surface erosion across Aotearoa, New Zealand. Rates reflect gross soil losses and do not account for sediment transport, redeposition, or the fate of such soil loss.

Table 4
Summary of soil lost from surface erosion across Aotearoa, New Zealand.

	Soil loss	Erodible area	Soil loss yield	Proportion
	t y^{-1}	ha	t ha^{-1} y^{-1}	%
North Island	5,561,548-16,033,451	10,821,705	0.51-1.45	34-55%
South Island	10,925,378-13,156,474	13,582,149	0.80-0.95	45-66%
New Zealand	16,486,926-29,189,926	24,403,854	0.68-1.17	100%

Thus, natural grasslands are not comparable equivalents of pastoral grasslands are typically characterized by shallower terrain and improved soil quality that support high pasture production. Further, higher soil losses from natural grasslands relative to pastoral grasslands cannot be used to infer that grazing reduces surface erosion.

Grasslands with native woody biomass exhibited negligible rates (Fig. 11) and proportions (<2%) of soil loss from surface erosion across the catchments (Fig. 12). Forested lands also contributed very minor proportions of surface erosion, apart from the Rangitaiki catchment which contained >80% forested land cover. In both land classes, the reduced soil losses were primarily the result of dense vegetation cover in mitigating soil loss, as we found no differences in slope, soil quality or rainfall for such land classes. This is supported by mechanistic understanding of vegetation's ability to mitigate surface erosion through rainfall interception, root cohesion, surface matting effects, and reducing the volume and rate of overland flow.

Despite having moderate rates of soil loss at local scales (0.09–5.01 t ha^{-1} y^{-1}), annual and perennial crops represented a small proportion of surface erosion at catchment scales because they occupied a small fraction of land cover. Together, annual and perennial crops represented ~1.4–2.3% of the land area and generally contributed ≤5% of soil loss via surface erosion.

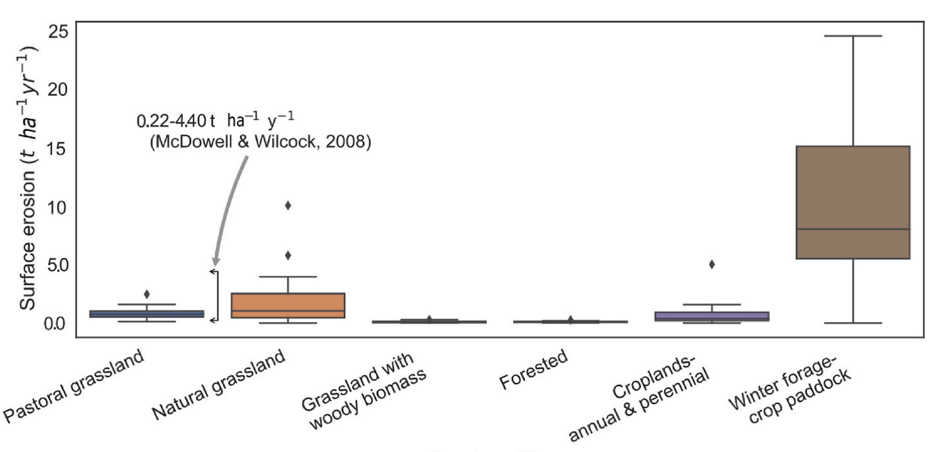

Fig. 11. Surface erosion for erodible landuse classes found across 22 catchments of Aotearoa, New Zealand. Surface erosion rates from grazed pastoral grasslands (leftmost boxplot) fell within the range of rates measured in literature (McDowell and Wilcock, 2008). Forage crop paddocks exhibited the highest rates of surface erosion (right, brown), reflecting exposed and degraded soil conditions from treading and grazing. High rates for natural ungrazed grasslands (orange, second from left) reflected low soil quality and high slope angles underlying such land, similar to previous studies (Panagos et al., 2015c). Grasslands with woody biomass and forested lands exhibited surface erosion rates on par with natural rates of soil generation and denudation (Montgomery, 2007).

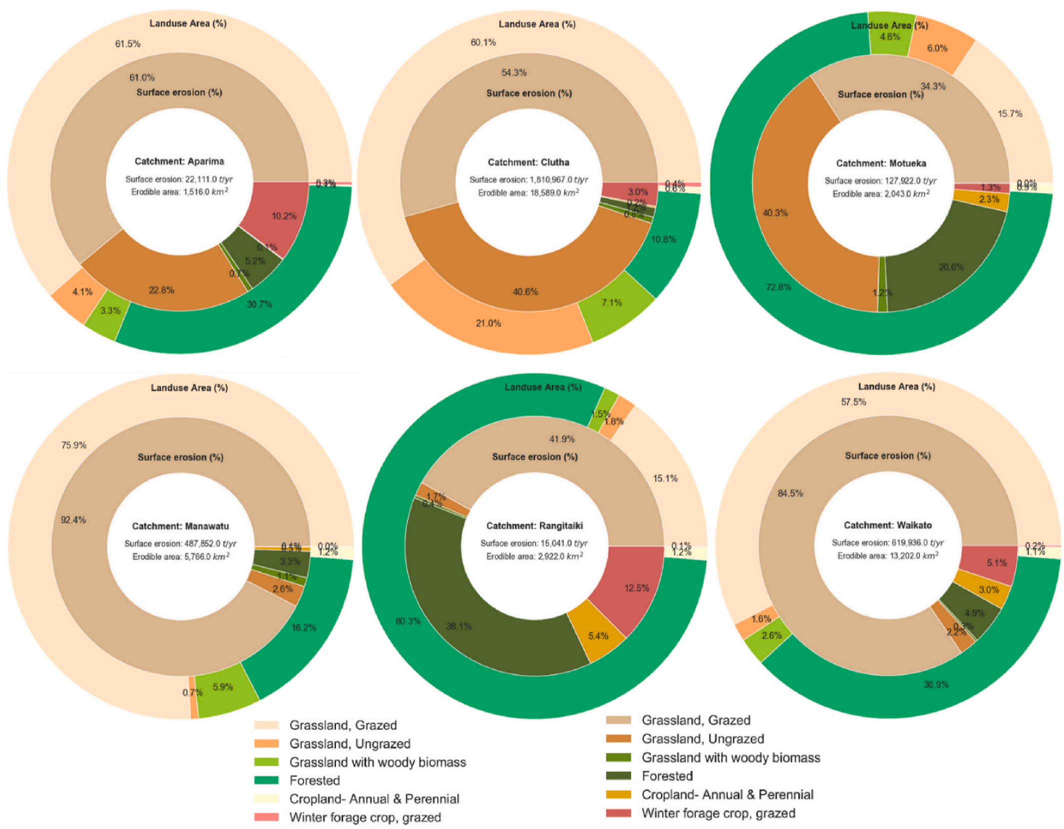

Fig. 12. Relative proportions of land use/cover and surface erosion from all erodible lands for six catchments across New Zealand with characteristics. The outer ring with light-shaded colors reflects the proportion of land use. The darker hues within the inner ring show the relative amounts of surface erosion for each land use/cover class. (Top) The Aparima, Motueka and Clutha were chosen as examples of South Island catchments containing a diverse range of landuse activity. (Bottom) The Manawatu, Rangitaiki, and Waikato were chosen as examples representing some North Island catchments.

4. Conclusions and future directions

We present the first national-scale model of soil lost via surface erosion that accounts for the impacts of grazing and treading on ground cover and soil erodibility (Donovan and Monaghan, 2021) using a RUSLE modelling framework that is capable for modelling alternative grazing management scenarios to understand the impacts to soil loss and catchment water quality. Modelled soil losses herein exhibited favorable alignment with spatially-explicit comparisons of field measurements, lake sediment cores, and chemical fingerprinting measurements of soil losses across a variety of land uses. Soil losses via surface erosion across Aotearoa, New Zealand, may reach 29.2 Mt y^{-1}, representing a loss of \$20M – \$110M annually.

Apart from forests and grasslands with woody biomass, annual soil losses were generally higher than natural rates of soil production and loss expected from native landscapes (Hancock et al., 2020; Montgomery, 2007). The effect of land management was most pronounced for intensive winter-forage crop paddocks, which exhibited high rates (11 t ha^{-1} y^{-1}) and highest relative proportions of annual soil loss (Figs. 11 and 12) across the 22 catchments analyzed. This reflected the combined effects of negligible residual ground cover following grazing, soil treading damage to wet soils, and relatively high slopes. Such areas thus represent the highest cost-benefit ratio for reducing soil losses across New Zealand, where increasing post-grazing residual ground cover could greatly enhance soil retention, thereby reducing the gap between rates of soil production and erosion (Hancock et al., 2020; Montgomery, 2007). Comparing modelled soil loss with downstream sediment yields from the Mahurangi Estuary (Supplementary Material, Appendix E) and subcatchments to the Manawatu river (Fig. 9) suggested that surface erosion could account for up to 24–32% of sediment yield over timescales sufficiently long to allow 100% sediment delivery. Future work should aim to model sediment transport and deposition to better resolve the proportion of soil losses from surface erosion that reach waterways.

Lastly, we demonstrate that topography has a dual role in impacting soil loss from surface erosion; directly through slope steepness and length, and indirectly through strong orographic effects that explained up to 50% variance of rainfall erosivity. Future work should further explore the relationship between elevation and rainfall erosivity at varying spatial scales to determine the scale at which orographic effects diminish in importance relative to broader weather patterns. Further, by incorporating expected climate change impacts to rainfall quantity and intensity, the framework herein could be used to project seasonal and annual soil loss scenarios.

Funding

This work was funded by the New Zealand Ministry for Business, Innovation and Employment's Our Land and Water National Science Challenge (Toitū te Whenua, Toiora te Wai), contract C10X1901 [Phase 2] / C10X1507 [Phase 1] as part of the Whitiwhiti Ora Land Use Opportunities research programme, supporting holistic decision-making to improve future landscapes of Aotearoa, New Zealand.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

Many thanks are extended to all those who contributed to the development of this research. Specifically, Ross Monaghan, Peter Pletnyakov, Richard Muirhead, Hugh Smith, David Houlbrooke, and Dan Sun for their expertise, technical support, and critique that helped improve the quality of work presented herein. A special thanks to Desneiges Murray for her statistical, graphing, and personal support of the work throughout its various iterations. We extend appreciation to Lucy Burkitt, Fiona Curran Cournane, Andrew Tait, and Wei Hu for sharing data and materials that helped quantify uncertainty of this model and promote transparent data sharing. Lastly, we thank the people of Aotearoa, New Zealand, who continue to support the conservation and restoration of the land; ngā mihi nui ki a koutou, he waka eke noa.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envsoft.2021.105228.

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