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Mark A. Nearing

*USDA-ARS Southwest Watershed Research Center*

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# 22

## Soil Erosion and Conservation

Mark A. Nearing

USDA-ARS Southwest Watershed Research Center, Tucson AZ, USA

### 22.1 The problem

Accelerated soil erosion induced by human activities is the principal cause of soil degradation across the world. The main culprit behind the problem is agriculture, and at stake is the long-term viability of the agricultural production capacity of the planet. Barring major unknown scientific advances in the future, and if soil erosion and population growth remain unchecked from their current rates, humanity will eventually lose the ability to feed itself. Another significant problem associated with soil erosion is off-site sediment pollution. Costs associated with the delivery of sediment to streams and other water bodies worldwide are huge (e.g. Pimentel, 1995). This chapter will focus on models of soil erosion as they are used for purposes of soil conservation. In particular, we focus here exclusively on soil erosion by water (see also Chapter 15). Models of other agricultural erosion processes, such as wind erosion and tillage erosion, are certainly important, but they will not be addressed here.

Models can be used in conservation work for three primary purposes: (a) to help a land owner or manager choose suitable conservation practices from among alternatives, (b) to make broad-scale erosion surveys in order to understand the scope of the problem over a region and to track changes in erosion over time, and (c) to regulate activities on the land for purposes of conservation compliance.

In selecting or designing an erosion model, a decision must be made as to whether the model is to be used for onsite or offsite concerns, or both. On-site concerns are generally associated with degradation or thinning of the soil profile in the field, which may become a problem

of crop-productivity loss. Conservationists refer to this process as *soil loss*, referring to the net loss of soil over only the portion of the field that experiences net loss over the long-term. Areas of soil loss end where net deposition begins. Off-site concerns, on the other hand, are associated with the sediment that leaves the field, which we term here *sediment yield*. In this case, we are not necessarily concerned with the soil loss, or for that matter the amount of sediment deposited prior to leaving the field, although estimation of both of these may be used to estimate sediment yields. Ideally, a model will compute soil loss, deposition and sediment yield, and thus have the capability to address both on-site and off-site issues.

Data variability and model uncertainty are two related and important issues associated with the application of erosion models. Data from soil-erosion plots contain a large amount of unexplained variability, which is an important consideration for using erosion data to evaluate soil-erosion models, as well as for interpreting erosion data. This variability is due both to natural causes and measurement errors. When comparing measured rates of erosion to predicted values, a portion of the difference between the two will be due to model error, but a portion will also be due to unexplained variance of the measured sample value from the representative, mean value for a particular treatment.

Knowledge of variability in soil-erosion data, however, is somewhat limited, although recent studies have enlightened us to some degree. Only one experimental erosion study to date has been conducted with a sufficient number of replicated erosion plots to allow an in-depth analysis of variability. Wendt *et al.* (1986) measured soil erosion rates on 40 cultivated, fallow, experimental plots located

in Kingdom City, MO, USA in 1981. All of the 40 plots were cultivated and in other ways treated identically. The coefficients of variation for the 25 storms ranged from 18% to 91%, with 15 of the storms falling in the range of less than 30%. The more erosive storms tended to show the lesser degree of variability. Of the 15 storms with mean erosion rates of greater than  $0.1 \text{ kg m}^{-2}$  ( $1.0 \text{ Mg ha}^{-1}$ ), 13 showed coefficients of variation of less than 30%. The results of the study indicated that 'only minor amounts of observed variability could be attributed to any of several measured plot properties, and plot differences expressed by the 25 events did not persist in prior or subsequent runoff and soil-loss observations at the site.'

Ruttimann *et al.* (1995) reported a statistical analysis of data from four sites, each with five to six reported treatments. Each treatment had three replications. Reported coefficients of variation of soil loss ranged from 3.4% to 173.2%, with an average of 71%. The authors concluded by suggesting 'as many replications as possible' for erosion experiments.

Nearing *et al.* (1999) studied erosion variability using data from replicated soil-loss plots from the USLE database. Data from replicated plot pairs for 2061 storms, 797 annual erosion measurements, and 53 multi-year erosion totals were used. They found that the relative differences between replicated plot pair measurements tended to decrease as the magnitude of the measured soil loss increased. Using an assumption that soil-loss magnitude was the principal factor for explaining variance in the soil-loss measurements, the authors were able to calculate the coefficient of variation of within-treatment, plot-replicate values of measured soil loss. Variances between replicates decreased as a power function ( $r^2 = 0.78$ ) of measured soil loss, and were independent of whether the measurements were event, annual, or multiyear values. Values of the coefficient of variability ranged from nearly 150% for a soil loss of  $0.1 \text{ kg m}^{-2}$  to as low as 18% or less for soil loss values greater than  $10 \text{ kg m}^{-2}$ . One important question for scientists is: 'How do we know when an erosion model is working adequately?' Given that the data are highly variable, when we ask the question about how well a model works, the answer is not so simple. One cannot just compare the model output to an erosion rate. One must simultaneously ask the question: 'How variable is nature?'

Risse *et al.* (1993) applied the Universal Soil Loss Equation (USLE) to 1700 plot-years of data from 208 natural runoff plots. Annual values of measured soil loss averaged  $3.51 \text{ kg m}^{-2}$  with an average magnitude of prediction error of  $2.13 \text{ kg m}^{-2}$ , or approximately 60% of

the mean. Zhang *et al.* (1996) applied the Water Erosion Prediction Project (WEPP) computer-simulation model to 290 annual values and obtained an average of  $2.18 \text{ kg}^{-2}$  for the measured soil loss, with an average magnitude of prediction error of  $1.34 \text{ kg}^{-2}$ , or approximately 61% of the mean. In both cases the relative errors tended to be greater for the lower soil loss values. Given these results and others from similar types of studies (Liu *et al.*, 1997; Rapp, 1994; Govers, 1991), the question may be asked: are the predictions 'good enough' relative to measured data? What is an acceptable and expected level of model prediction error?

One manner in which we can address this problem is to think of the replicated plot as the best possible 'real-world, physical model' of soil erosion. As such, one might further consider that the physical model represented by the replicate plot represents essentially a 'best case' scenario in terms of erosion prediction, which we can use as a baseline with which the performance of erosion prediction models might be compared. Using, as discussed above, data from natural runoff plots from the USLE plot database, Nearing (2000) suggested a basis for an erosion-model evaluation method using the idea of the replicate plot as a physical model of the replicated plot. He suggested that if the difference between the model prediction and a measured plot-data value lies within the population of differences between pairs of measured values, then the prediction is considered 'acceptable'. A model 'effectiveness' coefficient was defined for studies undertaken on large numbers of prediction versus measured data comparisons. The method provides a quantitative criterion for taking into account natural variability and uncertainty in measured erosion-plot data when that data is used to evaluate erosion models.

Nearing (2000) outlines the specific procedures for how erosion-model evaluation can be done in the presence of data uncertainty. The method is straightforward, but requires some detail in the computations. Using similar arguments with the erosion-plot replicate data, but using a slightly less complex analysis, we can achieve a rule-of-thumb measure of model validity simply by looking at the coefficient of determination for the regression line between measured and predicted soil-loss values. Using measured soil-loss data pairs from 3007 storms (overlapping with some of the same data used in the previously mentioned studies) Nearing (1998) obtained a coefficient of determination between measured and predicted soil loss of 0.77. One certainly would not expect, (on uncalibrated data) to obtain results between model predictions and measured data substantively better than

this, and for all practical purposes expectations of fit must be less. In the study by Risse *et al.* (1993) using the USLE and 1700+ plot years of data, the overall coefficients of determination were 0.58 for annual values and 0.75 for annual average soil loss data. In the study of Zhang *et al.* (1996), the WEPP model was applied using data from 4124 storm events, the coefficients of determination were 0.36 for the individual storms, 0.60 for annual values, and 0.85 for annual average soil-loss values. The observation that the fit improves from storm to annual to average annual predictions reflects the trend that data variability decreases with increasing soil-loss magnitudes, as discussed above.

Given that we know, based on the data from erosion plots, that soil erosion is highly variable, and then using the information on variability to set limits on the ability of models to predict soil-erosion rates, the question then becomes one of utility. Is the model accurate enough to solve our problems? We will address this question later in this chapter. But first we need to look at the models themselves, and look at an example of how an erosion model might be used to solve a problem.

## 22.2 The approaches

Erosion models used in applications for conservation planning fall into two basic categories: empirical and process-based. Undoubtedly the prime example of an empirically based model is the USLE, which was developed in the United States during the 1950s and 1960s (Wischmeier and Smith, 1965, 1978). This equation has been adapted, modified, expanded, and used for conservation purposes throughout the world (e.g. Schwertmann *et al.*, 1990; Larionov, 1993).

The USLE was originally based on statistical analyses of more than 10 000 plot-years of data collected from natural runoff plots located at 49 erosion research stations in the United States, with data from additional runoff plots and experimental rainfall-simulator studies incorporated into the final version published in 1978 (Wischmeier and Smith, 1978). The large database upon which the model is based is certainly the principal reason for its success as the most used erosion model in the world, but its simplicity of form is also important:

$$A = RKLSCP \quad (22.1)$$

where  $A$  ( $\text{t ha}^{-1} \text{a}^{-1}$ ) is average annual soil loss over the area of hillslope that experiences net

loss,  $R$  ( $\text{MJ mm h}^{-1} \text{ha}^{-1} \text{a}^{-1}$ ) is rainfall erosivity,  $K$  ( $\text{t hr MJ}^{-1} \text{mm}^{-1}$ ) is soil erodibility,  $L$  (unitless ratio) is the slope-length factor,  $S$  (unitless ratio) is the slope-steepness factor,  $C$  (unitless ratio) is the cropping factor, and  $P$  (unitless ratio) is the conservation-practices factor. Terminology is important here. Note first that the USLE predicts soil loss (see discussion above) and not sediment yield. Secondly, the word *erosivity* is used to denote the driving force in the erosion process (rainfall in this case) while the term *erodibility* is used to note the soil resistance term. These two terms are not interchangeable. Thirdly, the model predicts *average annual soil loss*: it was not intended to predict soil loss for storms or for individual years. Conservationists often describe the predictions as long term, whereas from the geomorphic perspective the predictions would be referred to as medium term (Govers, 1996).

The units of the USLE appear rather daunting as written (Equation 22.1), but become somewhat clearer with explanation. The units were originally written, and are still used in the United States, as Imperial, but conversion to metric is generally straightforward (Foster *et al.*, 1981). The key to understanding the dimensional units lies with the definition of rainfall erosivity and the concept of the *unit plot*. Wischmeier (1959) found for the plot data that the erosive power of the rain was statistically best related to the total storm energy multiplied with the maximum 30-minute storm intensity. Thus we have the energy term (MJ) multiplied by the intensity term ( $\text{mm h}^{-1}$ ) in the units of  $R$ , both of which are calculated as totals per hectare and per year. The unit plot was defined as a standard of 9% slope, 22.13 m length<sup>1</sup>, and left fallow (cultivated for weed control). The  $K$  value was defined as  $A/R$  for the unit plot. In other words, erodibility was the soil loss per unit value of erosivity on the standard plot. The remaining terms,  $L$ ,  $S$ ,  $C$  and  $P$  are ratios of soil loss for the experimental plot to that of the unit plot. For example, the  $C$  value for a particular cropped plot is the ratio of soil loss on the cropped plot to the value for the fallow plot, other factors held constant.

The USLE reduced a very complex system to a quite simple one for purposes of erosion prediction. There are many complex interactions within the erosional system, which are not, and cannot be, represented within the USLE. We will illustrate a few of these interactions below.

<sup>1</sup>Most of the early erosion plots were 1.83 m (6 feet) wide. A length of 22.13 m (72.6 feet) and a width of 1.83 m (6 feet) resulted in a total area of 1/100 of an acre. Prior to the days of calculators and computers this was obviously a convenient value for computational purposes.

On the other hand, for the purposes stated above for which an erosion model is used, the USLE has been, and still can be, very successful. This issue is also discussed below in more detail.

The USLE was upgraded to the Revised Universal Soil Loss Equation (RUSLE) during the 1990s (Renard *et al.*, 1997). This is a hybrid model. Its basic structure is the multiplicative form of the USLE, but it also has many process-based auxiliary components. It is computer based, and has routines for calculating time-variable soil erodibility, plant growth, residue management, residue decomposition, and soil surface roughness as a function of physical and biological processes. The RUSLE also has updated values for erosivity ( $R$ ), new relationships for  $L$  and  $S$  factors which include ratios of rill and interrill erosion, and additional  $P$  factors for rangelands and subsurface drainage, among other improvements. The RUSLE has the advantage of being based on the same extensive database as is the USLE, with some of the advantages of process-based computations for time-varying environmental effects on the erosional system. It still has the limitations, however, in model structure, which allows only for limited interactions and interrelationships between the basic multiplicative factors of the USLE (Equation 22.1).

Various process-based erosion models have been developed since the mid-1990s, including EUROSEM in Europe (Morgan *et al.*, 1998), the GUEST model in Australia (Misra and Rose, 1996), and the WEPP model in the United States (Flanagan and Nearing, 1995). We will focus here on the example of the WEPP model, largely because it is the technology most familiar to the author.

The WEPP profile computer model includes seven major components, including climate, infiltration, water balance, plant growth and residue decomposition, surface runoff, erosion, and channel routing for watersheds. The climate component of the profile computer model (Nicks, 1985) generates daily precipitation, daily maximum and minimum temperature, and daily solar radiation based on a statistical representation of weather data at a particular location. The climate model has been tested for erosion and well parameterized for the United States (Baffaut *et al.*, 1996). The infiltration component of the hillslope model is based on the Green and Ampt equation, as modified by Mein and Larson (1973), with the ponding time calculation for an unsteady rainfall (Chu, 1978). The water balance and percolation component of the profile model is based on the water balance component of SWRRB (Simulator for Water Resources in Rural Basins) (Williams and Nicks, 1985; Arnold *et al.*, 1990), with some modifications for improving estimation of

percolation and soil evaporation parameters. The plant-growth component of the model simulates plant growth and residue decomposition for cropland and rangeland conditions. The residue- and root-decomposition model simulates decomposition of surface residue (both standing and flat), buried residue, and roots for the annual crops specified in the WEPP User Requirements (Flanagan and Livingston, 1995) plus perennial crops of alfalfa and grasses. Surface runoff is calculated using a kinematic wave equation. Flow is partitioned into broad sheet flow for interrill erosion calculations and concentrated flow for rill erosion calculations. The erosion component of the model uses a steady-state sediment continuity equation that calculates net values of detachment or deposition rates along the hillslope profile (Nearing *et al.*, 1989). The erosion process is divided into rill and interrill components where the interrill areas act as sediment feeds to the rills, or small channel flows. The model is applicable to hillslopes and small watersheds.

Because the model is based on all of the processes described above, and more, it is possible with WEPP to have an enormous array of possible system interactions represented in the simulations. Just to name a very few examples, slope-length and steepness effects are functions of soil consolidation, surface sealing, ground residue cover, canopy cover, soil water content, crop type and many other factors. Ground residue cover is a function of biomass production rates, tillage implement types, residue type, soil moisture, temperature and solar radiation, previous rainfall, and many other factors. Rill-erosion rates are a function of soil-surface roughness, ground cover, consolidation of the soil, soil physical and chemical properties, organic matter, roots, interrill erosion rates, slope, and runoff rates, among other factors. The lists continue *ad infinitum*. These are interactions that simply cannot be represented with an empirical model. The WEPP is a very complex model in this sense.

The disadvantage of the process-based model is also the complexity of the model. Data requirements are huge, and with every new data element comes the opportunity to introduce uncertainty, as a first-order error analysis would clearly indicate. Model-structure interactions are also enormous in number, and with every structural interaction comes the opportunity for error, as well (see also Chapter 15). In a sense, the goal in using the process-based model is to capture the advantages of the complexity of model interactions, while gaining the accuracy and dependability associated with the simpler empirically based model. This goal can be achieved, and was achieved with the WEPP model, using a combination

of detailed sensitivity analyses and calibration of the model to the large database of natural runoff-plot information used to develop the USLE and RUSLE. Without the tie between model and database, and without knowledge of the sensitive input variables so as to know where to focus efforts, turning a complex model such as WEPP into a useful conservation tool would not be possible. Thus, in a sense, even though WEPP routines are process-based descriptors of various components of the erosional system, ultimately the model must be empirically based on the same type of data as was used to develop the USLE and RUSLE, along with additional experimental data collected specifically for WEPP.

### 22.3 The contributions of modelling

The accuracy of the three models introduced above has been tested using measured soil loss data from plots. We mentioned above the study by Risse *et al.* (1993) using the USLE and 1700+ plot-years of data, and the study of Zhang *et al.* (1996) of the WEPP model using data from 4124 storm events. The data of Risse *et al.* (1993) was also applied to the RUSLE model with very similar levels of accuracy as obtained with the USLE (Rapp, 1994). These three models all produced essentially equivalent levels of accuracy for prediction of soil loss, and the level was somewhat less than the level of fit obtained with the 'best-case' replicate plot-model discussed above. The results suggest that we have approached with these models the maximum level of possible soil-loss accuracy for ungauged, uncalibrated sites.

This result does not imply, however, that the three models are equivalent in usage. RUSLE has certain advantages over the USLE because its database and internal relationships have been expanded beyond that of the USLE for particular applications such as rangelands in the western United States and no-till cropped lands in the eastern United States. The data comparisons reported in the three studies above included no rangeland data and very little no-till data, so these advantages were not apparent from those studies. The USLE may have advantages in other applications. In areas where data are few, or computations need to be kept simple, the USLE has distinct advantages over both RUSLE and WEPP.

Another category of differences between the models is the type of information provided, rather than the accuracy of the information. The USLE provides essentially only average annual soil loss over the area of the field experiencing net loss. The RUSLE also provides only average

annual values of erosion, however, it provides estimates of off-slope sediment delivery in addition to estimates of on-slope soil loss. The RUSLE can also provide estimates of certain auxiliary system variables, such as residue amounts and crop yields. The WEPP model provides a massive array of system information to the user, if such information is desired. The model predicts both on-site soil loss and off-site sediment delivery, including ephemeral gully erosion, which neither USLE nor RUSLE attempts to predict. Sediment-delivery information includes not just the amount of sediment yield, but the particle-size distribution information for that sediment, which can be important in terms of chemical transport by sediment. The WEPP also provides a detailed description of the spatial and temporal distributions of soil loss, deposition, and sediment yields, both along the hillslopes and across the watershed. Auxiliary system information from WEPP is enormous, and is available on a daily basis. Information includes soil-water content with depth, surface residue amounts and coverage in both rill and interrill areas separately, buried residue and root masses, canopy cover and leaf area index, evapo-transpiration rates, soil surface roughness, soil bulk density, changes in hydraulic conductivities of the soil surface layer, changes in soil erodibility with consolidation and surface sealing, crop biomass and yields, subsurface interflow of water, tile drainage, and surface runoff amounts and peak rates, among others.

The USLE, RUSLE and WEPP (or other process-based models) constitute a complementary suite of models to be chosen to meet the specific user need. To illustrate this idea, we will take a look at recent applications of the USLE and WEPP to address the question of the potential impact of climate change on erosion rates in the United States. As we will see, we are able to use the USLE to provide certain information that WEPP simply cannot provide because of the restrictions of model complexity, and we are able to use the WEPP model in way where only the complex model interactions will provide us the information we want regarding system response.

In the first study we used the RUSLE R-factor to estimate the potential changes during the next century for rainfall erosivity across the whole of the United States, southern Canada, and northern Mexico. In this case, we do not want to become embroiled in the subtle differences between effects of various soils, slopes, cropping systems, and other system variables. Instead, we are looking for the primary effects over regions. With the USLE and RUSLE we can do this, because RUSLE uses an R-factor that was derived from a wide array of plot conditions, and it is not interdependent with the other system variables. Statistical

relationships have also been developed, as we shall see, between general precipitation data and erosivity. If we attempted to conduct such a broad-scale study with the WEPP model, we would quickly find ourselves with complicated sets of analyses, which we would then need to compose back to the general trends that RUSLE and the USLE provide directly. There would also be a data problem in this case, because WEPP requires certain details of precipitation that are not available from the global circulation models used to predict future climate change.

In the second study we review here, the objective was to determine the specific effects of changes in rainfall erosivity that might occur as a function of changes in the number of rain days in the year versus erosivity changes that are expected to occur when precipitation amounts per day and associated rainfall intensities change. In this study, the USLE and RUSLE would have been largely ineffective, because these changes are related to process changes within the system which USLE and RUSLE do not take into account. We shall see that in this case the detailed process interactions within WEPP enable us to see some quite interesting and important system interactions which significantly impact the results.

### 22.3.1 Potential changes in rainfall erosivity in the United States during the twenty-first century

Soil-erosion rates may be expected to change in response to changes in climate for a variety of reasons, including, for example, changes in plant biomass production, plant residue decomposition rates, soil microbial activity, evapo-transpiration rates, soil surface sealing and crusting, as well as shifts in land use necessary to accommodate a new climatic regime (Williams *et al.*, 1996). However, the direct, and arguably the most consequential, effect of changing climate on erosion by water can be expected to be the effect of changes in the erosive power, or erosivity, of rainfall. Studies using WEPP (Flanagan and Nearing, 1995) have indicated that erosion response is much more sensitive to the amount and intensity of rainfall than to other environmental variables (Nearing *et al.*, 1990). Warmer atmospheric temperatures associated with potential greenhouse warming of the earth are expected to lead to a more vigorous hydrological cycle, with the correspondent effect of generally more extreme rainfall events (IPCC, 1995). Such a process may already be taking place in the United States. Historical weather records analyzed by Karl *et al.* (1996) indicate that since 1910 there has been a steady increase in the area of the United States

affected by extreme precipitation events (>50.8 mm in a 24-hour period). According to statistical analyses of the data, there is less than one chance in a thousand that this observed trend could occur in a quasi-stationary climate. Karl *et al.* (1996) also observed in the weather records an increase in the proportion of the country experiencing a greater than normal number of wet days.

Atmosphere-ocean global climate models (see Chapter 9) also indicate potential future changes in rainfall patterns, with changes in both the number of wet days and the percentage of precipitation coming in intense convective storms as opposed to longer duration, less intense storms (McFarlane *et al.*, 1992; Johns *et al.*, 1997).

Rainfall erosivity is known to be strongly correlated with the product of the total energy of a rainstorm multiplied by the maximum 30-minute rainfall intensity during a storm (Wischmeier, 1959). The relationship first derived by Wischmeier has proved to be robust for use in the United States, and is still used today in the RUSLE (Renard *et al.*, 1997).

A direct computation of the rainfall erosivity factor, R, for the RUSLE model requires long-term data for rainfall amounts and intensities. Current global circulation models do not provide the details requisite for a direct computation of R-factors (McFarlane *et al.*, 1992; Johns *et al.*, 1997). However, the models do provide scenarios of monthly and annual changes in total precipitation around the world. Renard and Freimund (1994) recently developed statistical relationships between the R-factor and both total annual precipitation at the location and a modified Fournier coefficient (Fournier, 1960; Arnoldus, 1977), F, calculated from monthly rainfall distributions.

The example study that we want to examine here was conducted by Nearing (2001), who used the erosivity relationships developed by Renard and Freimund (1994) to estimate the potentials for changes in rainfall erosivity in the United States during the twenty-first century under global climate-change scenarios generated from two coupled atmosphere-ocean global climate models. The two coupled atmosphere-ocean global climate models from which results were used were developed by the UK Hadley Centre and the Canadian Centre for Climate Modelling and Analysis.

The most current UK Hadley Centre model, HadCM3 (Gordon *et al.*, 2000; Pope *et al.*, 2000; Wood *et al.*, 1999), is the third generation of atmosphere-ocean global climate models produced by the Hadley Centre. It simulates a 1% increase in greenhouse gases for the time period studied, as well as the effects of sulphate aerosols. The model also considers the effects of the minor trace gases CH<sub>4</sub>,

N<sub>2</sub>O, CFC-11, CFC-12, and HCFC-22 (Edwards and Slingo, 1996), a parameterization of simple background aerosol climatology (Cusack *et al.*, 1998), and several other improvements over the previous Hadley Centre model, HadCM2. Results from the model are reported on a 2.5° latitude by 3.75° longitude grid.

The Canadian Global Coupled Model (CGCM1) (Boer *et al.*, 2000), is composed of an atmospheric component based on the model GCMII (McFarlane *et al.*, 1992) coupled with an ocean component based on the model GFDL MOM1.1 (Boer *et al.*, 2000). For the current study we used results from the simulation GHG+A1, which incorporated an increase of atmospheric concentration of greenhouse gases (GHG) corresponding to an increase of 1% per year for the time period studied, as well as the direct forcing effect of sulphate aerosols (Reader and Boer, 1998). The data from this model were presented on a Gaussian 3.75° by 3.75° grid.

Changes in rainfall erosivity for the two models were computed for two time intervals, 40 and 80 years. In the first case the values of erosivity from the 20-year period from 2040 to 2059 were compared to the period 2000–2019, and in the second case the values of erosivity from the 20-year period from 2080 to 2099 were compared to the period 2000–2019. Erosivity changes were computed in two ways: (a) as a function of change in average annual precipitation for the twenty-year periods using equations 11 and 12 from Renard and Freimund (1994), and (b) as a function of the Fournier coefficient for the twenty year periods using equations 13 and 14 from Renard and Freimund (1994).

The erosivity results calculated from the Hadley Centre model analyses indicated a general increase in rainfall erosivity over large parts of the eastern United States, including most of New England and the mid-Atlantic states as far south as Georgia, as well as a general increase across the northern states of the United States and southern Canada (see maps in Nearing, 2000). The Hadley Centre results also indicated a tendency for erosivity increases over parts of Arizona and New Mexico. Decreases in erosivity were indicated in other parts of the south-western United States, including parts of California, Nevada, Utah, and western Arizona. Decreases were also shown over eastern Texas and a large portion of the southern central plains from Texas to Nebraska.

The erosivity results calculated from the Canadian Centre for Climate Modelling and Analysis model also showed an increase in erosivity across the northern states of the United States, including New England, and southern Canada (see maps in Nearing, 2001). The Canadian

Centre model results also indicated a reduction in erosivity across much of the southern plains, again from Texas to Nebraska, but extending somewhat west of the corresponding area shown in the Hadley Centre results. The Canadian Centre model did not show consistent results for the south-eastern United States. Results of the computations using the annual precipitation (see maps in Nearing, 2001) indicate changes in parts of the southeast United States tending toward lower erosivity, corresponding to a tendency toward a decrease in the annual precipitation in that region. Results of the erosivity computations using the Fournier coefficient indicate the possibility of little change or increases over part of the region for the 80-year comparison (see maps in Nearing, 2001). Calculated increases in erosivity using the Fournier coefficient suggest a change in the distribution of rainfall patterns through the year.

Erosivity results calculated from the Canadian Centre for Climate Modelling and Analysis and the Hadley Centre models show major differences in the south-western United States, including California, Arizona, Nevada, and Utah. Whereas the Hadley Centre model results suggest a definite trend towards lower erosivity in this area, the Canadian Centre for Climate Modelling and Analysis model results suggest a definite, strong trend toward greater erosivity through the twenty-first century.

The amount of inconsistency in the calculations from the two methods of calculating erosivity trends was, for the most part, similar between the two models (Table 22.1). Overall, between 16 and 20% of the calculations resulted in negative values of the R-factor calculated from total annual rainfall, RP, when the R-factor calculated from the Modified Fournier coefficient, RF, was positive, or *vice versa*. For the cases where both RP and RF were large, i.e., greater than 10%, those percentages were much smaller, although 7.6% of the pairs were inconsistent in this case for the Canadian model results for the 80-year time interval (2000–2019 to 2080–2099). It is not out of the question to expect inconsistencies between results of RP and RF, since RP is based on total annual precipitation and RF is based on the monthly distributions of precipitation. Both relationships are statistically based, and we have no reason to favour one over the other.

One might expect a consistent trend for the change of erosivity as a function of time, and in general this was true (Table 22.2). In this case, the Canadian model exhibited more inconsistency as function of time when using the monthly precipitation values to calculate erosivity, though it was consistent temporally in terms of the erosivity calculated using the annual precipitation.

**Table 22.1** Percentages of map grid cells in which changes over time in erosivity values, RP, calculated using precipitation were inconsistent in sign with changes in the values of erosivity, RF, calculated using the Fournier coefficient.

Model scenario	Inconsistencies in erosivity between RP and RF			
	For all Data		Where also both $ RP $ and $ RF  > 10\%$	
	40-yr. interval	80-yr. interval	40-yr. interval	80-yr. interval
	(%)	(%)	(%)	(%)
HadCM3	17.2	22.2	1.0	1.5
CGCM1 HG+A1	17.4	19.4	0.7	7.6

**Table 22.2** Percentages of map grid cells in which changes over time in erosivity values calculated over the 40-year time interval were inconsistent in sign with changes in the values of erosivity calculated over the 80-year time interval.

Model scenario	Inconsistencies in erosivity between 40- and 80-year time intervals			
	For all data		Where both the 40 y. $ R $ and 80 yr $ R  > 10\%$	
	RP	RF	RP	RF
	(%)	(%)	(%)	(%)
HadCM3	22.2	15.2	1.5	1.0
CGCM1 HG+A1	7.6	23.6	0	5.6

The RF values tended to show a somewhat greater magnitude, in terms of the average of the absolute value of percent erosivity change, than did the RP values (Table 22.3). The difference between the two models in this regard was striking. The Canadian model indicated a much greater level of erosivity changes overall as compared to the Hadley Centre model (Table 22.3). Both models suggested erosivity changes which generally increased in magnitude from the 40-year to the 80-year comparison.

### 22.3.2 Effects of precipitation-intensity changes versus number of days of rainfall

Now we take a look at another study of the effects of precipitation changes on soil-erosion rates, but this time we use the WEPP model. As we mentioned above, historical weather records analysed by Karl *et al.* (1996) indicate that since 1910 there has been a steady increase in the area of the United States affected by extreme precipitation events as well as an increase in the proportion of the country

experiencing a greater than normal number of wet days. The results given by Nearing (2001) discussed above provide a broad view of expected changes in erosivity based on the statistical models, but an important question not addressed is the expected differences in erosivity that come about relative to rainfall intensity versus a simple increase in the average number of rain days in a year. Erosion is not linearly proportional to rainfall intensity (Wischmeier and Smith, 1978; Nearing *et al.*, 1990).

Pruski and Nearing (2002) recently performed computer simulations to obtain estimates of potential runoff and soil-loss changes as a function of precipitation changes. In particular they studied the different responses of the erosional system to changes in precipitation as they occurred with changes in rainfall intensities, including the amount of rainfall that occurs on a given day of rain, versus responses to changes in simply the average number of days of rain. Assessments were made using WEPP for several combinations of geographic locations, soils, crops, and slopes. Geographic locations included West

**Table 22.3** Average magnitudes (absolute values) of erosivity change calculated.

Model scenario	Average magnitude of change			
	40-yr. interval		80-yr. interval	
	RP (%)	RF (%)	RP (%)	RF (%)
HadCM3	11.8	22.5	15.9	20.9
CGCM1 HG+A1	23.4	29.1	53.4	58.3

Lafayette, IN, Temple, TX, and Corvallis, OR. Soils were sandy clay loam, silt loam, and clay loam. Crops included grazing pasture, corn and soybean rotation, winter wheat, and fallow. Slopes were 3, 7, and 15%. Three scenarios of precipitation changes were considered: (a) all precipitation change occurring as number of days of rainfall, (b) all precipitation change occurring as amount of rainfall in a given day, and (c) half of the precipitation change occurring from each source. Under these scenarios, and using the climate generator for WEPP, changes in the number of days of rainfall does not influence rainfall intensity, whereas changes in the amount of rainfall on a given day increases the duration, peak intensities, and average intensities of rain. Levels of changes considered in each case were approximately zero,  $\pm 10\%$ , and  $\pm 20\%$  of total precipitation, with the same relative proportion of precipitation for the year maintained as a function of month.

Erosion rates changed much more with changes in the amount of rainfall per precipitation event, which also implies changes in the rainfall durations and intensities for the events. When total precipitation in this case was increased 10% in this case, soil loss increased an average of 26%. Realistically, we can expect that any changes in precipitation will come as a combination of both changes in the number of wet days as well as in changes in the amount and intensities of rainfall. As we discussed earlier, historical changes in rainfall over the past century have occurred in both of these terms (Karl *et al.*, 1996). For the combined case of both changes in wet days and changes in rainfall per day, Pruski and Nearing (2002) found that erosion responded intermediate to the two extremes. For a 10% increase in total precipitation, simulated erosion increased an average of 16%.

The average results for the combined case of changes in both number of days of precipitation and changes in amount of rain per day from the study of Pruski and Nearing (2002) are similar to those for the empirical relationship proposed by Renard and Freimund (1994)

between erosivity and total annual precipitation for the RUSLE model as discussed above. Using Renard and Freimund's first equation for erosivity results in a 17% change as a function of a 10% change in total annual precipitation. However, it is important to note that regardless of this fact, obtaining the broad-scale information on erosivity change similar to the information we obtained from the study discussed in the previous section (Nearing, 2001) would have been extremely difficult using WEPP.

Now let's look at some of the details of the results from the WEPP erosivity study. Greater amounts and rates of runoff, other factors being equal, will generally tend to cause an increase in erosion. Increased runoff causes increased energy of surface flow, which increases the detachment capability and the sediment transport capacity of the flow. Interrill erosion also increases with increased rain.

The simulation results of Pruski and Nearing (2002) showed a general increase in soil loss with increase in precipitation, and *vice versa* (Table 22.4), however, the changes were generally not as great as for runoff (Table 22.5). One major reason for the difference between the sensitivity results for runoff and those for soil loss is related to biomass production. Both runoff and soil loss are sensitive to biomass, but soil loss is more so. Soil loss is affected by plant canopy, which reduces the impact energy of rainfall; by crop residues, which protect the soil from raindrop impact and reduce rill-detachment rates and sediment-transport capacities; and from subsurface roots and decaying residue, which mechanically hold the soil in place and provide a medium in which micro-organisms can live. Thus, the increase of biomass production with increased rainfall tends to counteract to some degree the increased erosivity of the rain. This argument is supported by the results of the simulations for fallow conditions in comparison to the other treatments. The sensitivity values for the three precipitation scenarios for fallow conditions average 1.63 for soil loss and 1.55 for runoff. Thus fallow

**Table 22.4** Sensitivities of changes in soil loss to changes in average annual precipitation. Sensitivity values are calculated as the ratio of the percent change in soil loss to the percent change in precipitation. Values represent averages for all simulation runs associated with the soil, crop, slope, or location listed in the first column. Values greater than zero indicate that soil loss increases with increased annual precipitation. A value of greater than one indicates a greater percentage change in soil loss than the percentage change in precipitation.

Scenarios	Normalized sensitivity of soil loss to changes in average annual precipitation		
	Change in number of wet days	Change in amount of rain per day	Combined changes in Both
Silt loam soil	0.90	2.45	1.72
Sandy loam soil	0.89	2.60	1.82
Clay soil	0.79	2.10	1.46
Grazing pasture	1.02	2.66	1.96
Fallow	0.95	2.22	1.71
Corn and soybean	0.70	2.46	1.48
Wheat winter	0.77	2.18	1.50
S-shape (0%–3%–1%) 40 m	0.92	2.47	1.71
S-shape (0%–7%–1%) 40 m	0.84	2.40	1.67
S-shape (0%–15%–1%) 40 m	0.82	2.27	1.61
West Lafayette, IN	0.74	2.35	1.56
Temple, TX	0.88	2.10	1.50
Corvallis, OR	0.92	2.69	1.93
Overall average	0.85	2.38	1.66

was the only crop treatment for which the sensitivities for runoff were less than for soil loss.

The difference between a sensitivity of 0.95 for soil loss and 1.06 for runoff for the fallow scenario of change only in the number of days of rainfall (Tables 22.4 and 22.5) can be explained in terms of surface sealing and consolidation processes. Surface sealing and consolidation occur as a function of rainfall amount in nature and in the WEPP model (Flanagan and Nearing, 1995), so that any increase in rainfall will increase soil resistance to erosion via consolidation. This process also acts as a feedback effect, similar to the effect of rainfall-enhanced biomass growth, which partially offsets the impact of the increased rainfall on erosion and explains the lesser sensitivity of 0.95 for soil loss as compared to 1.06 for runoff.

The soil-loss-sensitivity value for fallow conditions for the scenario of change in amount of rainfall per day was greater (2.22) than that for runoff (1.99), whereas for the other crops the trend was reversed (Tables 22.4 and 22.5). Although the effects of surface sealing and consolidation, as discussed above, are present in this case, that effect is apparently superseded by yet another process when rainfall amounts and intensities per day are increased. These processes were related to rill and

interrill soil-detachment processes. Interrill erosion rates are represented in the WEPP model as proportional to the rainfall intensity and the runoff rate (Flanagan and Nearing, 1995), which are relationships based on experimental data (Zhang *et al.*, 1996). Both of these variables increase with increased rainfall intensity, so the effect of increased rainfall intensity on interrill erosion is greater than unity. Rill erosion also occurs as a threshold process. Rill detachment occurs proportional to the excess shear stress of water flow above the threshold critical shear stress of the soil, rather than to the shear stress of the flow itself. The overall effect is that the sensitivity of the rill erosion rate to runoff rate will be somewhat more than unity, other factors remaining constant. The effect is not present in the precipitation scenario of changes in the number of rainfall days because in that case, the average runoff rate is essentially not changing, but rather only the frequency of runoff events changes.

These are only a portion of the interactions discussed by Pruski and Nearing (2002) that were evident in the results of this study, but they provide a flavour of the types of information that the process-based model provides, which the empirical model cannot address. Hopefully the above discussions of these two model application will

**Table 22.5** Sensitivities of changes in runoff to changes in average annual precipitation. Sensitivity values are calculated as the ratio of the percent change in runoff to the percent change in precipitation. Values represent averages for all simulation runs associated with the soil, crop, slope, or location listed in the first column. Values greater than zero indicate that runoff increases with increased annual precipitation. A value of greater than one indicates a greater percentage change in runoff than the percentage change in precipitation.

Scenarios	Normalized sensitivity of runoff to changes in average annual precipitation		
	Change in number of wet days	Change in amount of rain per day	Combined changes in both
Silt loam soil	1.32	2.57	2.00
Sandy loam soil	1.31	2.80	2.17
Clay soil	1.15	2.17	1.75
Grazing pasture	1.54	3.09	2.41
Fallow	1.06	1.99	1.60
Corn and soybean	1.32	2.51	1.97
Wheat winter	1.21	2.43	1.91
S-shape (0%–3%–1%) 40 m	1.32	2.59	2.03
S-shape (0%–7%–1%) 40 m	1.29	2.49	1.98
S-shape (0%–15%–1%) 40 m	1.23	2.42	1.91
West Lafayette, IN	1.16	2.61	1.94
Temple, TX	1.19	2.25	1.73
Corvallis, OR	1.50	2.64	2.23
Overall average	1.28	2.50	1.97

provide the reader with a sense of how each type of model might be used to advantage depending upon the desired application.

## 22.4 Lessons and implications

At the start of this chapter we listed three primary uses for soil erosion models: (a) to help a land owner or manager choose suitable conservation, (b) to make broad-scale erosion surveys in order to understand the scope of the problem over a region and to track changes in erosion over time, and (c) to regulate activities on the land for purposes of conservation compliance. Let's look at each of these goals in turn.

Choosing how to manage land, from the practical perspective, is often a matter of choosing between an array of potential options. Often, therefore, what we need to know is not necessarily the exact erosion rate for a particular option to a high level of accuracy, but rather we want to know how the various options stack up against one another. We may certainly be interested to have a general quantitative idea of the erosion rate, but for purposes of land management, it is not critical. Choosing

which model to use then becomes a matter of (a) what type of information we would like to know, and (b) what information (data) we have for the particular site of application. We know from our discussions above that the USLE provides only estimates of average annual soil loss on the portion of the field that experiences a net loss of soil. If we have an interest in offsite impacts, then we probably want to choose either RUSLE, which will provide us with a rough idea of the sediment leaving the profile, or WEPP, if we want more comprehensive sediment-yield information or if we are modelling a small watershed. If we have an interest in obtaining other, auxiliary information about our choice of management strategy, such as soil moisture or crop yields, we might also decide to use WEPP. On the other hand, if data are limited for the situation to be modelled, then the USLE might be the best option in any case, and one would be forced to move to other options for assessing information not supplied by the USLE. At the current time most applications of WEPP are possible in the United States because of the availability of soil, climate and crop information, but in other areas this might not be the case.

Making broad-scale erosion surveys in order to understand the scope of the erosion problem over a region

and to track changes in erosion over time can be done with any of the models discussed above. Often a statistical sampling scheme is used to take random points over the area of interest, and to apply the erosion model to each point (USDA, 1996). In this case, too, we are not so concerned about the individual prediction for each point of application, but rather the ability of the model to predict overall averages of soil loss in a quantitatively accurate manner. While we know that none of these models will necessarily predict erosion for a particular site to the quantitative level of accuracy we would like to see for survey assessment purposes (Nearing, 2000), each of the three models does predict the averages for treatments quite effectively (Risse *et al.*, 1993; Rapp, 1994; Zhang *et al.*, 1996). As with the case discussed above, the issues related to choosing the correct model are related to the information desired and the available data.

Conservation compliance, governmental policy making, and regulation of land-users' actions follow the same guidelines as for the other two applications: information desired and data availability are again the keys to choice of model. In this case, however, the argument is often given, most often by the farmer who is being regulated, that if we know that there are uncertainties in the erosion predictions for individual applications, how can we be sure that his field is being evaluated accurately. The answer is, of course, that we cannot be sure. If the model predicts that the farmer's field is eroding at a rate in excess of what our society's policy indicates to be acceptable, the model could well be wrong for this particular field. This problem is really no different from that faced by insurance companies as they set rates for insurance coverage. My home may be more secure from the possibility of fire than my neighbour's home because I am more careful than my neighbour. But unless my home falls in a different category (for example, better smoke-alarm protection), I will not have much luck in going to my insurance company and asking for a lower payment rate. Likewise, if I am the farmer, I cannot expect to give a coherent argument for lower soil loss than the model predicts unless I conduct some practice, such as reduced tillage or buffers, which arguably reduces erosion.

Complexity and uncertainty are key issues relative to the development, understanding, and use of erosion models for conservation purposes. They are inevitable considerations because of the many complex interactions inherent in the erosional system as well as the enormous inherent variability in measured erosion data. These issues do not, however, prevent us from using models effectively for conservation planning. In fact, the scientific

evidence indicates that choice of models, which implies choice of model complexity, is more a matter of the type of information desired and the quality and amount of data available for the specific application. If our goal is to know to a high level of accuracy the erosion rate on a particular area of ungauged land, we cannot rely upon the models. Natural variability is too great, and uncertainty in predictions is too high (see Nearing *et al.*, 1999; Nearing 2000). For appropriate and common uses, such as those discussed above, models can be effective conservation tools.

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