



Research

Assessing temporal couplings in social–ecological island systems: historical deforestation and soil loss on Mauritius (Indian Ocean)

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ABSTRACT. Temporal couplings, such as historical interactions between deforestation and soil loss, are responsible for the current state of a wide range of ecosystem services of the social–ecological system on Mauritius. Islands are suitable study sites for understanding temporal couplings and telecouplings because of their: (1) clearly defined physical boundaries, (2) finite local resources, and (3) relatively short human history. Six well-documented historical deforestation maps, starting from the first colonization of Mauritius in 1638, were used as input parameters to model two scenarios of cumulative soil loss, with and without deforestation, using the revised universal soil loss equation in a geographic information system. The scenarios show that historical deforestation since 1638 has resulted in a cumulative soil loss that drastically exceeds soil loss under a natural baseline scenario without deforestation. The adopted method illustrates to what extent the current state of the soil of a social–ecological system is negatively affected by past human–environment interactions. We suggest that potential negative impacts on insular societies are mitigated by telecouplings such as food, fuel, and fertilizer imports.

Key Words: *deforestation; historical soil loss; islands; Mauritius; RUSLE; social–ecological systems; telecouplings; temporal couplings*

INTRODUCTION

Soils provide ecosystem services, such as water and nutrient retention, and form a substrate for agriculture (Millennium Ecosystem Assessment 2005, Braimoh and Vlek 2008). Soil loss, therefore, undermines the provisioning of many ecosystem services on which human well-being depends. Examples are negative impacts on agricultural productivity and soil quality by a reduction in nutrients, organic matter, soil biota, soil depth, infiltration rates, and water-holding capacity (Pimentel et al. 1995). Soil loss not only affects terrestrial ecosystem services, but marine ecosystem services as well. Increased sediment influx in the coastal zone leads to unwanted and increased turbidity and nutrient concentrations (Ramessur 2002, Fabricius 2005, Nagelkerken 2006, Liu et al. 2007b, Erftemeijer et al. 2012). Although natural soil loss is balanced by the process of soil formation, soils can be considered a limited natural resource because soil formation is a slow process (Montarella 2015). In general, rates of soil formation are closely correlated with erosion rates under native vegetation and the pace of erosion over geologic time, whereas erosion rates under agricultural practices are much higher (Montgomery 2007). Quantification of soil loss under native forest, therefore, provides a baseline to which human-induced soil loss can be compared.

Quantification of the responses of a social–ecological system (SES) to land-use change and deforestation is important for making informed decisions about balancing human demands and ecosystem functioning (DeFries et al. 2004). It is widely acknowledged that the current state of a SES is the outcome of patterns, processes, and decisions in the past (Foster et al. 2003, Kirch 2007, Rounsevell et al. 2012, Steen-Adams et al. 2015). The cumulative and evolving impacts of historical interactions within a SES on present and future conditions are termed legacy effects (Liu et al. 2007a), which suggests that the present-day soil is partly

degraded as a result of deforestation and agricultural practices in the past. However, the impact of soil degradation may remain unnoticed because of time lags between historical social–ecological couplings and their current effects (Liu et al. 2007b, Raudsepp-Hearne et al. 2010, Steen-Adams et al. 2015). These legacy effects and time lags are two important aspects for understanding temporal couplings in a SES (Liu et al. 2007a, Steen-Adams et al. 2015).

The current state of an island SES is also shaped by telecouplings; socioeconomic and environmental interactions with other SES over spatial distances (Liu et al. 2013, Eakin et al. 2014, Friis et al. 2016). These telecouplings, such as increased global economic connectivity, complicate the interplay between drivers and consequences of land-system change (Lambin and Meyfroidt 2011, Meyfroidt et al. 2013). Taking telecouplings into account helps to acknowledge that ecosystem services can be partly substituted in space by food and fuel imports and enables a decoupling of a society from its local ecosystem. As a result, time lags between soil degradation and its perceived societal impact may increase. It can, therefore, be hypothesized that the continued reliance on telecouplings allows for the development of a “soil debt,” which is a net decrease of the total soil stock in a region over time, that results when erosion rates exceed soil formation rates. The debt can be paid when conditions promoting soil recovery, such as reforestation, are restored.

Islands provide ideal conditions for understanding the complex historical couplings between humans and their natural environment because of three island characteristics (Kirch 1997, Fitzpatrick and Keegan 2007, Royle 2012, Warren et al. 2015): (1) clearly defined physical boundaries; (2) finite local resources, such as land, soil, water, and timber; and (3) for many islands, a relatively recent human colonization history. Many island

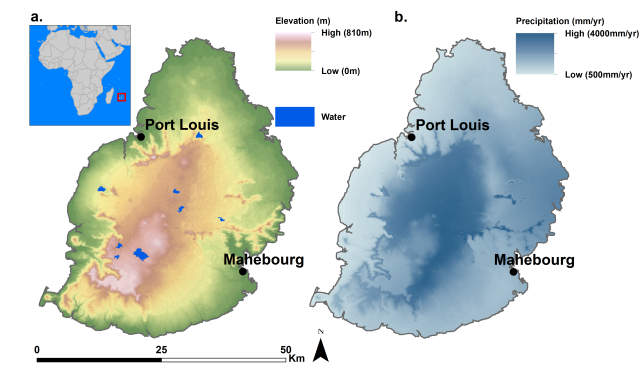
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ecosystems have been drastically transformed following human colonization (Strasberg et al. 2005, Caujapé-Castells et al. 2010). Additionally, human colonization can coincide with a substantial increase in erosion and sedimentation (Lepofsky et al. 1996, Kahn et al. 2015), resulting in drastic alteration of the landscape and ecosystem functioning. On Mauritius, human settlement is closely linked to forest fragmentation, fires, introduction of exotic species, and degradation of natural ecosystems (Cheke and Hume 2008, Rijsdijk et al. 2011, 2015, Florens et al. 2012, De Boer et al. 2013). These human impacts eventually affect the ecosystem's capacity to provide services to society. The introduction of exotic species, for example, has been shown to negatively influence supporting, provisioning, cultural, and regulating services (Vilà et al. 2010). The objective of this paper is to analyze the effects of temporal couplings on the current state of an island SES. In addition, we will assess how far the capacity of the soil to provide ecosystem services to Mauritian society is undermined. Mauritius forms an ideal study site for such an analysis because its social-economic and deforestation history are relatively well documented. This allowed for an historical analysis of human-environment interactions and the role of telecouplings. Temporal couplings on Mauritius were operationalized by quantifying the influence of historical deforestation on cumulative soil loss. Comparison of the cumulative soil loss resulting from deforestation to natural background variation yields insight into the degree of human impact on island ecosystems and environments (Swetnam et al. 1999, Connor et al. 2012). The history of human-environment interactions on Mauritius, including the role of telecouplings, is included in this comparison.

Historical human-environment interactions and telecouplings

The island Mauritius (1865 km²) is located in the Indian Ocean approximately 860 km east of Madagascar and measures roughly 50 by 60 km (Fig. 1). The climate on Mauritius has two seasons: a rainy summer (70% of annual rainfall) from November to April dominated by cyclone passage, and a dry winter from May to October (Nigel and Rughooputh 2010). The central plateau and the eastern part of the island receive the highest annual precipitation (Fig. 1).

Fig. 1. Elevation map of Mauritius (left panel) and annual precipitation averaged over 30 years (right panel). Left panel is based on data from Seul (1999) and Hill (2001), right panel is based on data from Padya (1984) and Mauritius Meteorological Services (2005).

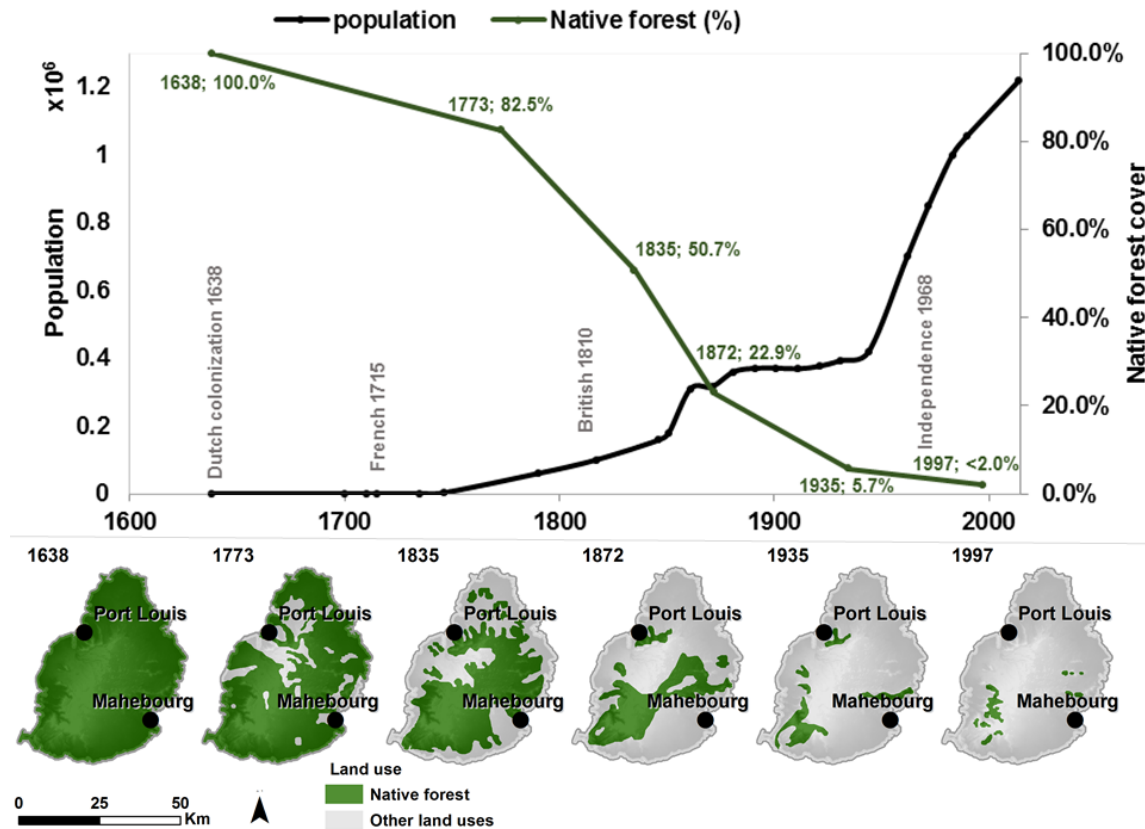


Mauritius was first colonized in 1638 by the Dutch, who used the island mainly as a refreshment station. A variety of crops were introduced, such as tobacco, sweet potato, and sugar cane (Brouard 1963). Domestic animals and deer were brought to the island, the giant tortoises that grazed the coastal areas were killed for their meat, and trees in the lowland coastal forests were felled to be exported. The first felling of trees occurred in the direct surroundings of the harbors near present-day Mahebourg (Grand Port District) and Port Louis (see deforestation maps in Fig. 2, lower panel). Although deforestation occurred at a slow pace and was localized (Griffiths and Florens 2006), nearly all accessible large black ebony (*Diospyros tessellaria*) trees were felled (Brouard 1963, Moree 1998). In this period, only a few hundred colonists permanently inhabited Mauritius under Dutch rule (Fig. 2, upper panel), and eventually, the island was abandoned in 1710.

After Mauritius was abandoned by the Dutch, it was claimed by France in 1715 and settled in 1721. Mauritius functioned as a trading post to supply French ships in the Indian Ocean with essential goods and services (Allen 1989). Mauritius's role as a commercial trading center greatly promoted local economic activity and laid the foundation for the formation of domestic capital (Allen 2008). Local merchants and seafarers made investments in agricultural land (Allen 2008), and some public land concessions were given out freely to settlers (Grove 1996). The local economy was primarily occupied with providing goods and services to the ships that made a stopover in the harbors, and the local government actively encouraged the cultivation of sugar, spices, and other crops (Allen 1989, 2008). Sugar cane became the preferred and dominant agricultural crop, and a plantation system was introduced. The French brought slaves from Madagascar, Mozambique, and West Africa to Mauritius primarily to work on the sugar cane plantations (Addison and Hazareesingh 1993). Deforestation continued, not only to make room for sugar cane plantations, but also to meet the growing timber demand for naval and construction purposes (Grove 1996).

The French plantation system became more institutionalized after Mauritius was conquered by the British in 1810 (Alladin 1986). This marks the beginning of a transformation from a trade-oriented economy toward a plantation economy, predominantly focused on sugar production for the British imperial market (Allen 2008). Slaves continued to form the primary workforce on the sugar cane plantations. By 1817, 80,000 slaves lived on Mauritius, out of a total population of nearly 100,000 (Lutz and Wils 1994, Teelock 1998). Major expansion of sugar cane occurred after 1825 when the British government allowed the import of Mauritian sugar and removed the tax penalty in Britain (Allen 1989). In addition, sugar consumption started to increase profoundly in Britain, which transformed sugar cane into a main cash crop (Addison and Hazareesingh 1993). Sugar exports became the predominant contributor to the Mauritian economy. After 1830, sugar contributed 85–90% to export earnings (Allen 2008); from that moment onward, almost all food and other consumer products were imported (Brookfield 1959). As a result of the intensification of sugar cane agricultural activities in the 19th century, deforestation increased drastically (Florens 2013). The agricultural regions expanded mainly in the accessible coastal areas, leaving the central areas covered with forest (Brookfield 1959). Starting in 1834, after the abolition of slave labor, the

Fig. 2. Population growth and deforestation on Mauritius since human colonization (upper panel). Population graph based on Koenig (1944), Titmuss and Abel-Smith (1960), Lutz and Wils (1994), and Statistics Mauritius (2015a), percentage of native forest cover from Florens (2013). The lower panel shows six timeslices indicating the areas where native forest was removed since colonization in 1638. The six maps are based on Vaughan and Wiehe (1937) and references therein, and Page and D'Argent (1997).



British brought indentured laborers from India to form the primary workforce on the plantations. From the 1870s onward, large sugar estates started selling their less productive land to smallholders, driven by labor shortages and a decreasing sugar price due to competition with European sugar beet (Addison and Hazareensingh 1993, Meisenhelder 1997). This period, referred to as the “Grand Morcellement,” led to the establishment of a large number of smallholder farms. By the early 1920s, almost half of the area under sugar cane was cultivated by smallholders of Indian origin (Brookfield 1959, Allen 2008). However, the land division was highly unequal; in 1946, there were 30 factory estates, 109 large estate planters, and 13,685 planters owning small plots of often marginal quality (Brookfield 1959).

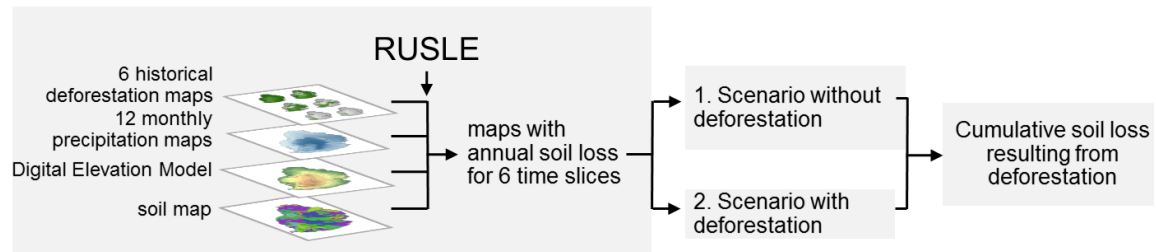
In 1968, the year of independence, sugar accounted for 93% of the nation’s exports (Meisenhelder 1997), whereas in 2014 this was only 3.85% (Mauritius Chamber of Commerce and Industry 2015). Whereas the area under sugar cane remained relatively stable at around 80,000 ha (43% of land area) between the year of independence and 1980, this area decreased rapidly in the last decades from 76,840 ha (41.5%) in 1990 to 50,687 ha (27.4%) in 2014 (Statistics Mauritius 2012, 2013a, 2014). During the 1970s

and 1980s, the Mauritian economy underwent major structural changes, with a rapid phase of industrialization, diversifying into two major activities, textiles and tourism (Ramessur 2002). With the diversification of the economy, vegetated areas on Mauritius have been converted to other land uses, mainly for urban developments like roads, hotels, and housing (Hammond et al. 2015). These economic changes were accompanied by rapid population growth (Fig. 2, upper panel). Today, the population of Mauritius has risen to 1,219,659 people (Statistics Mauritius 2015a). On top of this, tourist arrivals have doubled in the last two decades, from 486,867 in 1996 to 1,038,968 in 2014 (Statistics Mauritius 2013b, 2015b). Much of the food that is needed to feed the growing population is being imported. Both in volume and monetary terms, import of most foodstuffs such as cereals, fruits, nuts, vegetables, dairy, and meat products largely exceeds exports (Mauritius Chamber of Commerce and Industry 2015).

METHODOLOGY

To model historical soil loss scenarios, we adopted the widely applied revised universal soil loss equation (RUSLE; Renard et al. 1997). We use the RUSLE because it has successfully been applied in soil erosion studies on Mauritius before (Le Roux et

Fig. 3. Workflow illustrating the RUSLE-based analysis of cumulative soil loss for two scenarios of deforestation on Mauritius.



al. 2005, Nigel and Rughooputh 2012). By using six historical deforestation maps, 12 monthly precipitation maps, a digital elevation model (DEM), and a soil map as input parameters for the RUSLE, two cumulative soil loss scenarios were constructed (Fig. 3).

The first scenario (“scenario without deforestation”) represents a situation in which Mauritius would never have been colonized and the entire island remained covered with native forest. The soil loss in 1638 was used as the annual soil loss rate to calculate the cumulative soil loss for the scenario without deforestation. The second scenario, with deforestation, represents a situation in which forest is gradually converted to sugar cane. For this scenario, six historical deforestation maps showing native forest from the years 1638, 1773, 1835, 1872, 1935, and 1997 (Vaughan and Wiehe 1937, Page and D’Argent 1997) were digitized and georeferenced to construct input parameters for the RUSLE. The modeled soil loss for these timeslices was used as annual soil loss rates to reconstruct the cumulative soil loss for the scenario with deforestation. Linear interpolation was used to reconstruct soil loss rates for the intermediate years between the modeled timeslices. Comparison of both scenarios provides insight into the extent to which historical deforestation impacts cumulative soil loss because the timeframe and parameters for soil type, precipitation, steepness, etc. are the same in both scenarios. These cumulative soil loss scenarios provide a proxy of the legacy effects of historical deforestation.

We have automated the soil loss modeling workflow using the RUSLE in ArcGIS 10.1 model builder. The RUSLE (Renard et al. 1997) equals:

$$A = R \times K \times LS \times P \times C \quad (1)$$

where A is the computed soil loss ($t\ ha^{-1}\ year^{-1}$), R is the rainfall erosivity factor ($MJ\ mm\ ha^{-1}\ h^{-1}\ year^{-1}$), K is the soil erodibility factor ($t\ ha\ MJ^{-1}\ ha^{-1}\ mm^{-1}$), LS is the topographic factor (unitless), P is the support practice factor (unitless), and C is the cover management factor (unitless). Soil loss has been calculated on a monthly basis and is subsequently aggregated per year. An overview of the data sources where the RUSLE input parameters are derived from is presented in Table 1. Here, the transformation steps carried out to prepare the input data for the RUSLE are briefly discussed; a more extensive overview of the methodology is presented in Table A1.1. (Appendix 1). The RUSLE model and input data are shared in the supplementary materials (Appendix 2).

Table 1. Metadata and sources used to develop input maps for the RUSLE. A detailed description of data transformations to develop RUSLE factors can be found in Table A1.1 in Appendix 1.

Factor	Data source
R	Precipitation data from Mauritius Meteorological Services (2005) and Padya (1984).
K	Digitized vector map with spatial distribution of soil types from Willaime (1984).
LS	Soil erodibility factors from Nigel and Rughooputh (2012). Digital Elevation Model (DEM) of Mauritius based on the work of Seul (1999) and Hill (2001).
P	Land-use distribution for six timeslices from Vaughan and Wiehe (1937) and references therein, and Page and D’Argent (1997).
C	Support practice factor from Nigel and Rughooputh (2012). Land-use distribution for six timeslices from Vaughan and Wiehe (1937) and references therein, and Page and D’Argent (1997). Cover management factor from Nigel and Rughooputh (2012).

Rainfall is the driving factor for initiating sheet and rill erosion (Yang et al. 2003). R represents the kinetic energy of rainfall and was calculated by using a modified Fournier index (Arnoldus 1977, 1980). Precipitation data from Mauritius (Padya 1984, Mauritius Meteorological Services 2005) indicate that this index closely coincides with calculated R values for the climate stations Curepipe, Vacoas, and Plaisance on Mauritius (Atawoo and Heerasing 1997). K represents the soil loss specific to a soil type, measured on a standard plot. The soil map comes from Willaime (1984), erodibility factors have been extracted from Nigel and Rughooputh (2012). The LS factor combines slope length (L) and slope steepness (S). These factors are calculated from a 100 x 100 m resolution DEM, which is based on the work of Seul (1999) and Hill (2001). The geomorphological zonation of Saddul (2002) indicates that sedimentation is dominant on <3% slopes, which was accounted for in the model by deselecting slopes <3% from the soil loss classes. Another additional adaptation is that lakes were excluded from the flow length calculation using a Boolean operator and a digitized lake data set. The urban area of Port Louis has been masked as well. The values for P and C are representative for a particular land-use type and reflect local agricultural practice. Values for the support practice factor of two land-use types (sugar cane and native forest) on Mauritius are based on the work of Nigel and Rughooputh (2012).

Subsequently, these P and C values have been assigned to the land-use types in historical deforestation maps (Fig. 2, lower panel) georeferenced and digitized from Vaughan and Wiehe (1937) and Page and D'Argent (1997). For further details and parameter specific equations of the RUSLE, we refer to Arnoldus (1977), Wischmeier and Smith (1978), and Renard et al. (1997).

Some generalizations were necessary to quantify the effects of historical deforestation on soil loss during a period of nearly four centuries. Even though historical deforestation on Mauritius is relatively well documented, most generalizations stem from the limited detail of historical maps. First, although different forest types existed, no distinction is made. The native forest before human settlement consisted of dry palm woodlands on the leeward side of the island, inland semidry evergreen forest, and high-elevation wet forests (De Boer et al. 2013). Although Mauritius was almost entirely forested before human settlement (van der Plas et al. 2012), it is known that heath and marshy vegetation covered substantial areas (Vaughan and Wiehe 1937, De Boer et al. 2013). In addition, the coastal habitats were likely inhabited by a plant community of grasses and herbs grazed by the currently extinct giant tortoises *Cylindraspis inepta* and *C. triseriata* (Hansen et al. 2010). Second, in our model, all native forest is converted to sugar cane whereas it was also cleared to accommodate the growing population, for settlements, and for planting food crops. Regarding urban areas, approximately 5.5–9% of the present-day land area is occupied by manmade structures such as buildings, roads, etc. (Nigel et al. 2014, Hammond et al. 2015). Because the growth of urban areas throughout history is not accurately known, these areas are not included in the analysis, the only exception being the city of Port Louis. Soil loss under urban areas is usually considered to be zero (Nigel and Rughooputh 2010, 2012), although it might lead to increased runoff downstream. In addition, the soil protection of urban areas in the past was probably lower than today because it had less impervious surfaces. Ignoring most of the urban growth will likely lead to an overestimation of soil loss. Regarding food crops, the modeled soil loss will likely be underestimated because soil loss under vegetable crops is generally higher than under sugar cane (Le Roux et al. 2005, Nigel and Rughooputh 2012). From the 18th century onward, tea plantations were established in the wet uplands that were unsuitable for sugar cane. However, for most of the time, the crop only covered a small part of the island; until the 1960s, never more than 1.5% (Brookfield 1959). Today, tea covers 0.4–1.6% of the total land surface (Nigel et al. 2014, Statistics Mauritius 2014). Tea plantations have a lower erosion risk than sugar cane and vegetables because they are not periodically denuded at harvest (Nigel and Rughooputh 2010, 2012). Third, reforestation and establishment of timber plantations are not included in the analysis. Between 1880 and 1900, 1,600 ha (0.9% of total land surface) were planted with fast-growing tree species by the Forest Department (Brouard 1963). Timber plantations of mainly loblolly pine (*Pinus taeda*) were established in 1914, and large-scale slash pine (*Pinus elliottii*) plantations were established from 1932 onward (Brouard 1963). From 1950 to 1959, 200–400 ha (approximately 0.1–0.2% of the island) were planted annually with fast-growing species such as Eucalyptus (*Eucalyptus robusta*, *E. kirtoniana*, and *E. tereticornis*), pine trees (*Pinus elliottii* and *P. taeda*), and Araucaria (*Araucaria cunninghamii*). Today, 25.3% of the island is covered

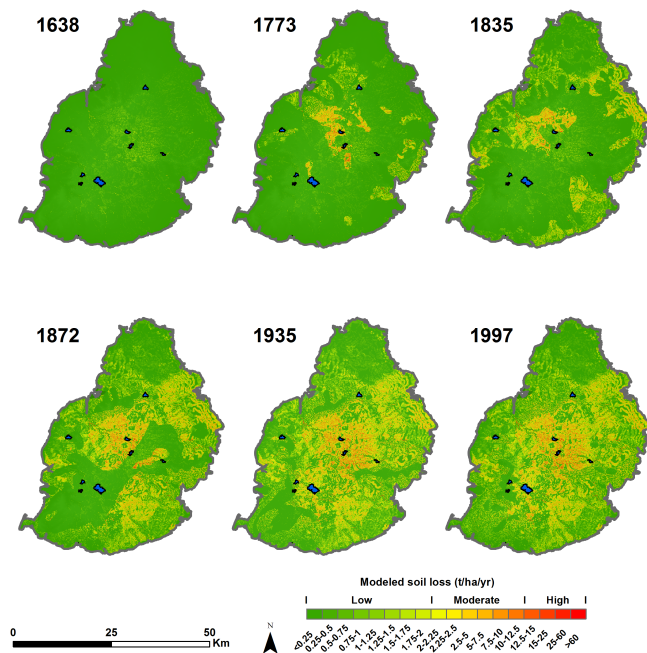
with “forests, shrubs and grazing lands” (Statistics Mauritius 2014), of which less than 2% consists of native forest (Florens 2013). The reason for not including nonnative forests and timber plantations in our analysis is that their exact location through time is not accurately known. Timber plantations do not provide the same soil protection as native forests because the trees are harvested every 7–10 years, when considering fast-growing tree species. It is often considered that 3 years after harvesting, the protective cover of timber plantations is comparable to that of a native forest (Lu et al. 2003). However, timber plantations have a much lower resistance than native forest to the heavy rains associated with cyclones that are common on Mauritius (Brouard 1963), increasing their erosion risk. Although soil disturbance under timber plantations is generally higher than native forest, it is lower than sugar cane because it has a longer cutting cycle. This leads to the fourth limitation: subannual and interannual changes in vegetation cover are not considered. Sugar cane is harvested annually by cutting the crop at ground level, after which the soil is left relatively bare before the crop has fully regrown again (Mardamootoo et al. 2013). Once every 7–8 years on average, the cane is replanted, for which the soil has to be tilled as well. The perennial character of sugar cane reduces erosion susceptibility under sugar cane compared with annually harvested crops (Mardamootoo et al. 2015). Although the practice of leaving crop residues on the field after harvesting is common, preharvest cane burning is still practiced as well, leaving the soil exposed and enhancing erosion. This relates to the fifth limitation: the support practice factor and cover management factor in the RUSLE are taken to be constant for a specific land-use type. As a result, differences between land owners (small vs. large growers) and related differences in sugar cane cultivation practices (e.g., manual vs. mechanized production) are not considered, whereas these may lead to differential landscape outcomes (Steen-Adams et al. 2015). During the first centuries of settlement, all sugar cane fields were harvested and planted manually. Today, large estates have mechanized their production process, but most small growers still do the planting and harvesting by hand. Throughout history, small growers often cultivated lands of inferior quality and produced less efficiently and less intensively compared with large growers (Brookfield 1959), which might lead to lower soil loss. Finally, the historical variation in precipitation and soil erodibility is not accurately known for the timescale of our analysis. Therefore, precipitation data for a 30-year time period was used to model soil loss. Soil erodibility based on a present-day soil map is used, although soil parameters, such as carbon content, soil thickness, and soil type are not static. In our model, soil formation and soil thickness are not considered, which might result in an overestimation of soil loss.

RESULTS

The influence of historical deforestation on soil loss in the different timeslices is clearly visible (Fig. 4). In 1638, when the entire island was covered by native forest, the modeled soil loss values for nearly the entire island are within the lowest modeled soil loss class (0–2 t/ha/yr; Fig. 4). As native forest is being converted to sugar cane in subsequent timeslices (1773, 1835, 1872, 1935, and 1997), several locations within the island fall within the moderate (2–12.5 t/ha/yr) and high (>12.5 t/ha/yr) modeled soil loss classes (Fig. 4). In 1997 (Fig. 4), areas in the center and northeastern parts of the island show high modeled

soil loss values. These areas are characterized by the highest amounts of annual precipitation (Fig. 1). Most lowland coastal zones and the western and northwestern part of the island have low rainfall values and low slope angles, which results in low to moderate modeled soil loss values.

Fig. 4. Modeled soil loss for six timeslices since colonization in 1638. For visualization purposes, the modeled soil loss has been classified into three classes: low (0–2 t/ha/yr), moderate (2–12.5 t/ha/yr), high (>12.5 t/ha/yr).

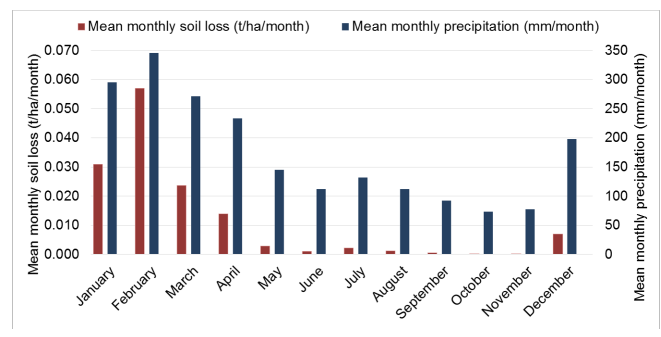
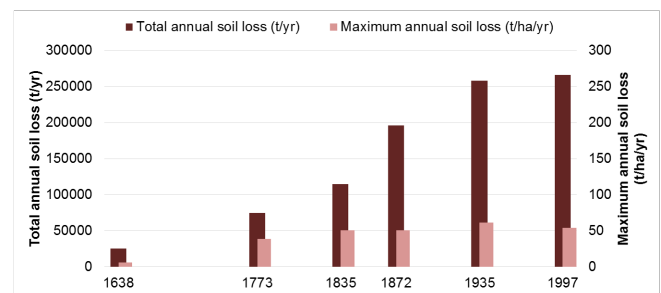


The maximum modeled soil loss for the 1638 timeslice with full native forest cover is 5.7 t/ha/yr (see upper graph of Fig. 5). When assuming a bulk density of 1,200 kg/m³ (Montgomery 2007, Nigel and Rughooputh 2012), it can be calculated that this maximum modeled soil loss value under native forest is equivalent to a soil loss of 0.47 mm/yr. Between 1835 and 1935, wet areas on the central plateau with high annual rainfall were deforested, which generated high modeled soil loss, with a maximum of 61.4 t/ha/yr in 1935, equating to a soil loss of 5.1 mm/yr (assuming a bulk density of 1,200 kg/m³). It is clear that monthly precipitation and modeled monthly soil loss per hectare are closely related (Fig. 5, lower graph). The graph shows that seasonal variability in rainfall drastically influences the magnitude of modeled soil loss in different months. As a result, months with extreme precipitation have a prominent influence on the modeled annual soil loss.

Based on the modeled annual soil loss in the six timeslices (Fig. 5, upper graph), the cumulative soil loss since the year 1638 for the whole island is reconstructed for two scenarios (Fig. 6). In the scenario without deforestation, the cumulative modeled soil loss until the year 1997 is 9.2 million tonnes (green line in Fig. 6). In the scenario with deforestation, the cumulative soil loss increased drastically during the various stages of deforestation on

Mauritius (orange line in Fig. 6) up to a total soil loss of 49.1 million tonnes in 1997. Between 1638 and 1773, the percentage of native forest decreased to 82.5% of the original cover. In 1773, the cumulative soil loss was already twice as high compared with the baseline scenario. By 1835, when almost half of the island was deforested, cumulative soil loss was 2.5 times higher than the baseline scenario. The cumulative soil loss increases more rapidly in the second part of the 19th century. By 1872, when approximately 75% of the island had been deforested, the cumulative soil loss exceeded the baseline scenario by a factor of 3.1. As the felling of native forest continued, the cumulative modeled soil loss in 1935 and 1997 exceeds the baseline scenario by a factor of 4.3 and 5.4, respectively.

Fig. 5. Total annual modeled soil loss for the whole island (t/yr) and maximum value of modeled annual soil loss within the whole island (t/ha/yr) for each of the six analyzed timeslices (upper graph) and mean monthly modeled soil loss (t/ha/month) for the 1638 timeslice and mean monthly precipitation (mm/month, lower graph).

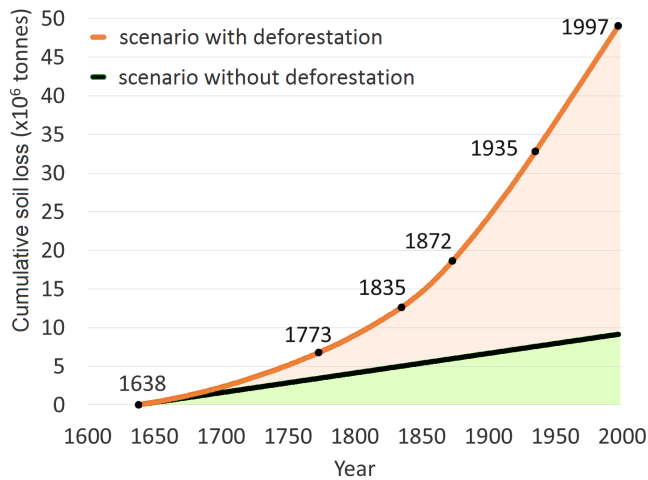


DISCUSSION

Temporal couplings

The cumulative soil loss scenarios provide an approximation of human impact relative to natural baseline erosion rates. Because of the limited detail and availability of historical data, some generalizations were necessary. Short-term fluctuations and the effect of extreme events are averaged out. However, such generalizations are inescapable for any study in which past human–environment interactions are modeled (Dearing et al. 2006, Fitzpatrick and Keegan 2007). Despite these generalizations, historical deforestation and soil loss over a period of nearly four centuries show some interesting patterns. Until the end of the 18th century, deforestation was limited to the easily accessible locations and areas close to the harbors. As a result,

Fig. 6. Cumulative modeled soil loss on Mauritius for a scenario with and without deforestation. To calculate cumulative soil loss for the scenario without deforestation, the soil loss in 1638 was used as the annual soil loss rate. In the scenario with deforestation, linear interpolation was used to reconstruct soil loss rates for the intermediate years between the modeled timeslices.



modeled soil loss during these early stages of deforestation was low. Deforestation and modeled soil loss started to accelerate in the first half of the 19th century when Mauritius was being transformed into a plantation economy. The general pattern that emerges on Mauritius shows that modeled soil loss increases over time and was highest during the most recent stages of deforestation. This temporal trend in soil loss is typical for the historical development of human impact on island ecosystems and environments, which starts gradually after initial colonization and accelerates as human populations grow and land use increases and intensifies (Rick et al. 2013). On Mauritius, during initial human settlement, lower fertile coastal regions were exploited first, whereas the less fertile higher grounds—especially those on steeper slopes—were exploited more recently. This might reflect a more general pattern on islands where settlements are mostly concentrated in lowland zones and high and steep areas are not permanently inhabited (Kirch 2007). Modeled soil loss resulting from deforestation drastically exceeds the natural variation in soil loss, leading to a growing soil debt. The soil debt only provides a general indication of how ecosystem services are influenced by deforestation. Our analysis does not show local details related to deforestation and soil loss. For example, actual impact of soil loss on provisioning services is closely related to the loss of topsoil, which typically has a higher fertility (Pimentel et al. 1995). Therefore, an initial erosion event in a certain locality has stronger impact on fertility than a younger erosion event under similar conditions. Although time lags can exist before the impacts of historical land-use change and deforestation become visible (Foster et al. 2003, DeFries et al. 2004), historical sources show that the impact of deforestation on soil fertility and droughts on Mauritius were already recognized during the 18th and 19th centuries (Brouard 1963, Grove 1996). For example, in a

government report dating from 1870, it is mentioned that “large-scale soil denudation has continued the work of destruction by deforestation. [...] Finally, the fertility of soils has been greatly affected” (Régnaud 1870 quoted in Brouard 1963). In recent years, 36% of the soils had a phosphorus deficiency (Mardamootoo et al. 2010), and average yield of sugar cane per hectare has been declining during the last decades (Umrit et al. 2014, Cheong and Umrit 2015). These findings agree with a study on Hawaii, where nearly four centuries of agricultural practices have significantly reduced soil fertility (Kirch 2007). On Mauritius, loss of soil fertility does not seem to be among the major causes of land abandonment; small planters that leave their land fallow, or shift from sugar cane to vegetable cultivation, are mainly motivated by social-economic factors (Lalljee and Facknath 2008). Aside from the loss in soil fertility, several other ecosystem services are impacted by soil loss. Soil loss leads to increased water runoff, lower infiltration rates, and lower water-holding capacity (Pimentel et al. 1995), which can affect vegetation growth. Offsite effects of erosion processes include siltation, loss of reservoir storage, and increased water treatment costs (Pimentel et al. 1995). These aspects are especially relevant for Mauritius given the island’s chronic water scarcity (Ramjeawon 1994) and its 74% dependency on external water resources (Hoekstra and Mekonnen 2012). These important impacts strongly suggest that the capacity of the soil to provide ecosystem services has been undermined.

Historical soil loss mitigation

Raudsepp-Hearne et al. (2010) raised the question why human well-being is increasing, while ecosystem services are degrading on a global scale. The analysis of temporal- and telecouplings on Mauritius provides an interesting perspective on how this apparent paradox is expressed on a local scale. Despite the negative impacts of historical deforestation and soil loss on ecosystem services, Mauritius is generally considered a success story considering its sustained economic growth and the well-being of its inhabitants (Sobhee 2009, Bunwaree 2014), although not all segments of the population benefited equally (Bunwaree 2014). Between 1980 and 2013, the human development index (HDI) of Mauritius has been increasing gradually; today, Mauritius holds the 63rd position of 187 countries and territories (United Nations Development Programme 2014). This seems to coincide with a decreasing dependency of the Mauritian economy on locally provided ecosystem services. Today, agriculture (including sugar), forestry, and fishing contribute only 3% to the gross domestic product (GDP), and the manufacturing of, respectively, foodstuffs and sugar contributes an additional 6.1% and 0.2% (Mauritius Chamber of Commerce and Industry 2015). Globally, land becomes an increasingly scarce resource, which leads to growing competition between different land uses (Lambin and Meyfroidt 2011). On the island of Mauritius, where all land is allocated and is scarce, three out of the four major Mauritian sugar cane companies have invested in property on the African continent (<http://www.omnicane.com/>, <http://www.alteogroup.com/>, <http://www.terra.co.mu/>). The latter illustrates that land-use change on islands cannot be understood in isolation, as Baldacchino (2004) rightfully notes, “Islands are not islands, in the sense that they are not closed unto themselves.” Since its colonization, Mauritius always held strong links and dependencies with other localities (Eriksen 1993), through its

dependency on slave labor, its imports of food and fertilizer, and for a long time, its sugar exports. It seems that throughout history, the negative impacts of deforestation and soil loss have been partly offset by a strong dependency on trade in global markets. For example, in the 1950s, 30% of the sugarcane lands on Mauritius had a phosphorus deficiency, after which phosphorus imports increased tenfold between 1955 and 1970 (Mardamootoo et al. 2010). Although the capacity of the soil to provide ecosystem services to society has been undermined as a result of temporal couplings, negative impact on the SES of Mauritius seems to be partly mitigated by telecouplings with other SES. However, a heavy reliance on fossil energy and fertilizer imports to mitigate the consequences of soil loss can likely not be sustained over the long term (Pimentel et al. 1995) and has both economic and environmental costs. Mitigating the impacts of soil loss by increasing the use of fertilizers can negatively influence other ecosystem services. On Mauritius, high phosphorus and nitrogen concentrations in runoff waters have been recorded (Ng Kee Kwong et al. 2002), leading to eutrophication in the coastal zone (Ramessur 2002), which might in turn negatively impact coastal fisheries and tourism. In addition, 50% of the energy consumption involved in sugar production on Mauritius is taken up by fertilizers (Ramjeawon 2004). Finally, fossil energy and phosphate rock, which are both used for fertilizer production, are finite (Cordell et al. 2009), which further stresses the need for Mauritius to pay back the soil debt. While paying this debt, those localities with highest soil erodibility should receive high soil conservation priority and, during periods of heavy rainfall, vegetation cover should be ensured (Nigel and Rughooputh 2010, 2012). Moreover, restoration approaches should take into account the degree to which the soil has already been degraded by temporal couplings such as historical land use and deforestation.

CONCLUSION

Our objective was to study the effects of temporal couplings on the current state of a SES. We operationalized these temporal couplings by focusing on the influence of historical deforestation on cumulative soil loss on Mauritius. Based on historical deforestation maps, we modeled soil loss for six timeslices since human colonization of Mauritius in 1638. We have compared the soil loss resulting from deforestation to a baseline scenario in which the entire island remained covered by forest.

We conclude that deforestation has led to a drastic increase in soil loss since first human settlement. Cumulative soil loss resulting from deforestation exceeds the natural soil loss baseline by more than five times. Such a soil loss is not sustainable over the long term when assuming soil formation rates to be in equilibrium with soil loss under native vegetation. It is further concluded that the cumulative soil loss undermines the capacity of ecosystems to provide services to society today and in the future, but that the negative consequences of soil loss are partly mitigated by telecouplings with SES abroad. The latter underpins that islands, as any locality, cannot be understood in isolation and that connections with other SES should be taken into account. Future research on insular SES should address that the current state of a SES emerges from the interaction between temporal couplings and telecouplings.

Responses to this article can be read online at:

<http://www.ecologyandsociety.org/issues/responses.php/9073>

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APPENDIX 1

Table A1.1. Overview of data transformations to develop RUSLE factors. All raster files have a cell size of 100x100 meters (1 hectare).

Factor	Input data	Transformation
R	Precipitation data from Mauritius Meteorological Services (2005) and Padya (1984).	<p>For the calculation of pi, the monthly rainfall over 30 years between 1951 and 1980 from 23 stations was interpolated at a 100m resolution. The interpolation was conducted per month (thus resulting in 30 x 12 = 360 maps).</p> <p>Five interpolation methods were evaluated: linear, inverse distance weighted, cubic spline, generalized additive modelling in combination with kriging, and radial basis functions. Elevation, aspect and gradient were evaluated as covariates, along with x- and y-coordinates, in the interpolations. Only elevation appeared to be a meaningful covariate.</p> <p>The performance of each interpolation method was measured by calculating the root mean squared error through leave-one-out cross-validation (Hastie et al. 2009). Through this evaluation, the interpolation by radial basis functions turned out to be the best performing technique (rmsq = 79.9 mm), followed by inverse distance weighted interpolation (rmsq = 114.6 mm). Linear interpolation was the method with the lowest performance.</p> <p>The results from the radial basis function interpolation were used as average monthly precipitation (Pi) to calculate a modified Fournier index (F). Usually, the EI30 rainfall erosivity index (Wischmeier and Smith 1978) is used but in absence of rainstorm intensity data, it was replaced by the modified Fournier index (Arnoldus 1977, 1980) which linearly correlates with the rainfall erosivity index (Ferro et al. 1999). The modified Fournier index was calculated as follows:</p> $F = \frac{\sum_{i=1}^{12} p_i^2}{P}$ <p>where Pi = average monthly precipitation (mm) and P is the average annual precipitation (mm).</p> <p>Subsequently, the modified Fournier index was used to calculate an R factor for Mauritius:</p> $R = rF^a$ <p>where <i>r</i> and <i>a</i> are location specific parameters. We have used <i>r</i> = 0.00302 and <i>a</i> = 1.9, following Arnoldus (1980), as cited by Le Roux et al. (2005).</p>

K	Digitized vector map with spatial distribution of soil types from Willaime (1984). Soil erodibility factors from Nigel and Rughooputh (2012).	K values assigned. Digitized vector map converted to raster (100x100m).
LS	Digital Elevation Model (DEM) of Mauritius from (Seul 1999) and Hill (2001).	L is the length factor, λ the slope length (m) and m is the slope-length exponent (m = 0.5 on slopes >5°, 0.4 on slopes between 3° and 5°, 0.3 on slopes between 1° and 3°, 0.2 on slopes <1°). Slope length is normalized for a unit plot of 22.13m. $L = \left(\frac{\lambda}{22.13} \right)^m$ Slope steepness is calculated differently for slopes < 9° and for slopes ≥9°: $\text{For } S < 9^\circ, S = 10.8 \sin\beta + 0.03$ $\text{For } S \geq 9^\circ, S = 16.8 \sin\beta - 0.50$
P	Land use distribution for six time slices from Vaughan and Wiehe (1937) and references therein, and Page and D'Argent (1997). Support practice factor from Nigel and Rughooputh (2012).	P values assigned. Digitized vector maps converted to raster (100x100m).
C	Land use distribution for six time slices from Vaughan and Wiehe (1937) and references therein, and Page and D'Argent (1997). Cover management factor from Nigel and Rughooputh (2012).	C values assigned. Digitized vector maps converted to raster (100x100m).

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Appendix 2. RUSLE GIS model and input data for Mauritius.

[Please click here to download file 'appendix2.zip'.](#)
