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Impacts of historical land use changes on erosion and agricultural soil properties in the Kali Basin at Lake Balaton, Hungary

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Abstract

Historical land use changes may have significant impact on erosion and agricultural soil properties, including soil degradation by acidification, nutrient leaching and organic matter depletion. The Kali Basin study area, a small catchment of high landscape value located in a national park at Lake Balaton, Hungary, with its historical agricultural records, together with the available unique historical land use data for the last 200 years, provides an opportunity to study and model impacts of historical land use changes on erosion and agricultural soil properties. Comparison of long-term land uses with present soil degradation indicator parameters showed that permanent arable land use has led to degradation of both the physical and chemical properties of soils in the Kali Basin. Application of the SEDEM/WATEM distributed erosion and sediment transport model showed that, despite the low overall sediment export from the catchment, land use changes introduced by property ownership and agricultural changes have decreased average soil erosion in the catchment but increased relative sediment export to Lake Balaton. This is due to changes in the land cover pattern that allow more sediment transported to the river system. The overall conclusion of this study is that besides the size and area proportion of land use types, land use pattern seems to be equally important in soil erosion and degradation processes, thus land use pattern is a key factor for landscape planning and development in the Kali Basin. A relationship between the sociological and agro-ecological reasons for the recorded land use changes is also shown in this study.

Keywords: Agriculture; Catchment; Erosion model; GIS; Land use change; Soil

1. Introduction

Agricultural land use is one of the most important factors that have shaped historic landscapes in Europe (Black et al., 1998; Correia, 2000; Bicik et al., 2001; Burgi and Russel, 2001). Increasing agricultural exploitation of landscapes and associated land use changes have often led to soil quality degradation and loss of soil by erosion. Protection of soil as an important non-renewable natural resource has been recognised as a priority in the European Union (COM(2002) 179), in part due to the results of scientific observations of soil degradation (Van Oost et al., 2000; Van Rompaey et al., 2002; Sun et al., 2003). In order to develop efficient strategies for the sustainable management of soil resources it is essential to understand and model processes that can lead to soil quality degradation due to land use practices. One way to investigate the effect of land use on soil quality is to analyse long-term records of land use and to study the impacts of land use changes on soil properties. The Kali Basin in Hungary with its historical agricultural record, together with the available unique historical land use data for the last 200 years, provides an opportunity to study and model impacts of historical land use changes on erosion and agricultural soil properties.

The Kali Basin is one of the areas with the longest agricultural history in Hungary. Vine growing on the slopes

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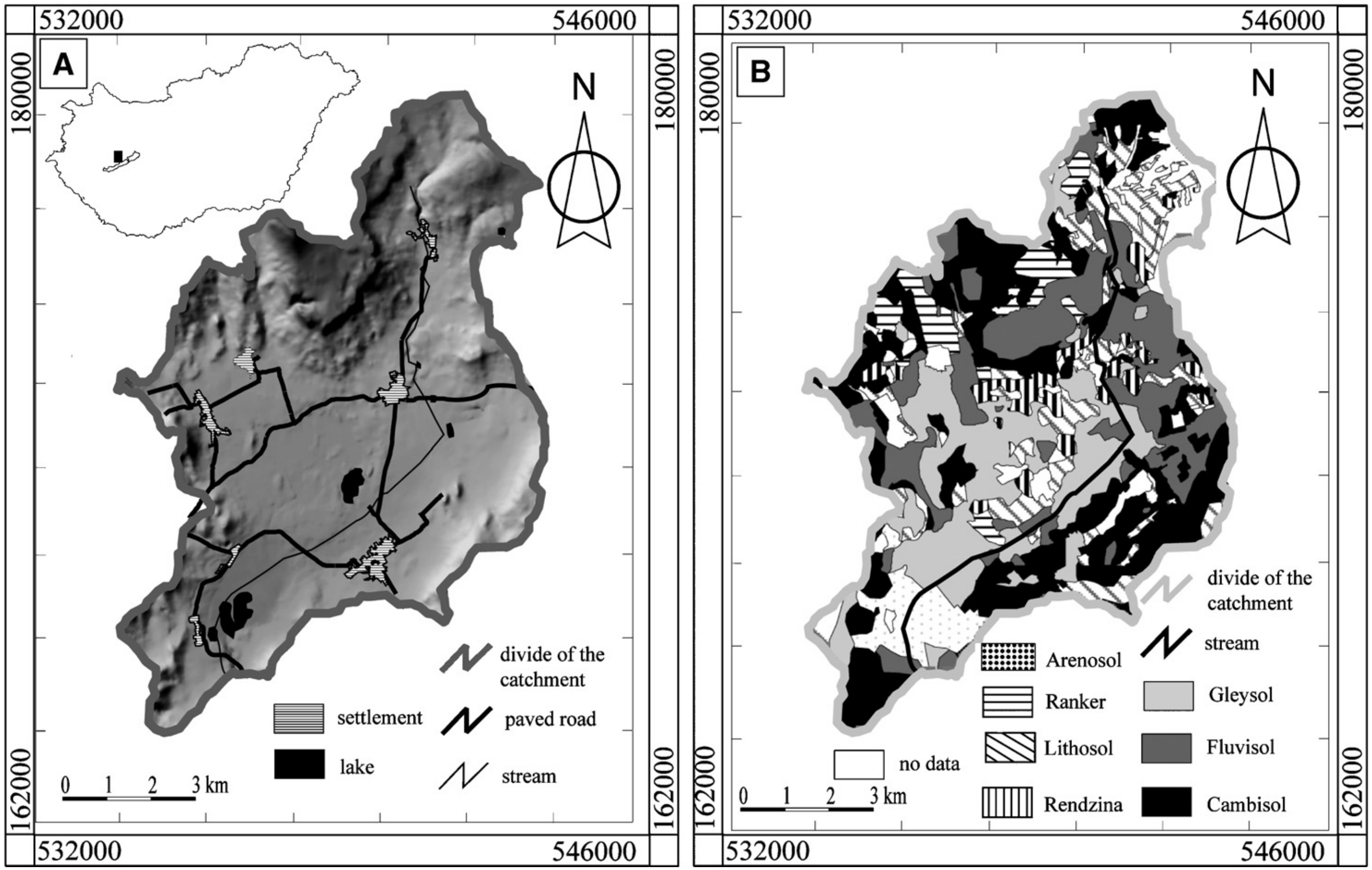

Fig. 1. The Kali Basin study area. A. Topography of the study area, also showing relief by a shaded relief model in the background. Inset: location of the study area at Lake Balaton in Hungary. B. Soil types in the study area.

bordering the basin has been ongoing since the Roman times, for almost 2000 years. Vineyards represent the most prominent landscape value in the basin. The Kali Basin became a protected Natural and Landscape Reserve in 1984 and it has been a part of the Balaton Highland National Park since 1997. The objective of long-term environmental management in the Kali Basin is, on one hand, to protect historic agricultural landscapes and, on the other hand, to develop an optimal land use pattern that minimises erosion and soil degradation.

The most dramatic land use changes in the study area were due to (1) sudden increase of number of cultivated parcels due to population boom to historic maximum in the 1930s, (2) change to centrally directed economy after the WWII and the following introduction of large-scale agricultural technologies, and (3) transition to market economy a decade ago. These land use and land management changes are typical in most of the Central and Eastern European New EU Member States and Candidate Countries (Lipsky, 1995) thus the study of land use change dynamics in this region is essential to understand legacy of the past and to develop appropriate tools for integration into the EU agricultural, soil and land management strategies.

The objective of this study was to investigate impacts of historical land use changes on soil erosion and agricultural soil properties in the Kali Basin study area, a small catchment located in a national park at Lake Balaton, Hungary (Fig. 1A). Sociological and agro-ecological reasons for the land use changes were also investigated in this study.

2. Materials

2.1. Soils and physiography of the Kali Basin

Permian red sandstone builds-up the southern hills while the basin is bordered by Pliocene basaltic volcanic masses in the north and west in the Kali Basin (Budai and Csillag, 1999). In the middle of the basin and in the east gently folded Triassic carbonate sediment series are exposed. Tertiary clastic sediments, primarily sand, fill the majority of the basin. Quaternary deposits consist of wetland sediments in the majority of the basin area, while slope-foot scree, alluvial sediments and Pleistocene loess in valleys are characteristic of the study area. Due to variable geological, morphological and hydrological conditions soils show a great variety and spatial diversity (Fig. 1B). Cambisol dominates as defined by the Middle and Southern Climatic Zone of Europe (Szaszine, 2000). Based on the available soil maps used in this study, cambisol(2) refers to a variation of typical cambisol (called cambisol(1) in this paper) that is characterised by clay accumulation in the lower part of the soil section. Rendzina and ranker soils are found over carbonate and basaltic rocks, respectively (Fig. 1B). On the slopes of surrounding hills lithosol and often fluvisol are found (Fig. 1B). The majority of the flat basin area is covered by gleysol defined by local surface and groundwater conditions. Based on the available soil maps used in this study, gleysol(2) refers to a variation of typical gleysol (called gleysol(1) in this paper) that is characterised by intensive organic matter accumulation in the upper part of the soil section due to flood waters (Szaszine, 2000).

The watershed has an area of 82.2 km^2 with elevation range between 104 m and 474 m a.s.l. Valley network of the study area is poorly developed (Jordan, 2003; Jordan et al., 2003) (Fig. 1A). Climatic conditions are typically of temperate continental character with long-term (50 years) annual mean temperature of 9.5–9.7 °C, 1960–1980 h of sunshine and with an annual mean precipitation of 650–680 mm.

2.2. Land use changes

There have been agricultural activities in the Kali Basin since the Roman times. A strong economic decline in the area was due to the Turkish occupation in the 17th century. Arable lands around settlements and vineyards on steep hill slopes became characteristic in the 18th century again, according to the First Military Survey Map published in 1784 (Fig. 2A). Terraces were developed for vine growing on the steep upper sections of hill slopes this time (Laposa, 1988). According to the Second Military Survey Map published in 1854, vineyards were cultivated also on the lower slopes at hill feet (Fig. 2B; compare to Fig. 1A). Vineyards were recultivated after the 1889–1891 filoxera disease that abolished about 90% of vineyards. However, vineyards on the steeper upper slopes with poor access were not recultivated and vine terraces have been taken over by natural forest since then (Szilassi, 2000). Population and extent of cultivated lands peaked in the 1930s in the Kali Basin (Fig. 2C). At the same time, there was a significant increase of arable lands in this period (Szilassi, 2001, 2003). Total area of vineyards decreased in this period due to economic reasons (Laposa, 1988). At the same time, increase in the number and decrease in the size of parcels have blocked agricultural development in the area.

After WWII, vineyards dropped back to the level of the filoxera disease (Fig. 2D). Due to the introduction of centrally directed economy, lands were collectivised and large-scale agricultural technologies were introduced. As a result, large homogeneous parcels developed, former wetlands were turned into arable lands and vineyards were recultivated but only at the lower hill slopes at the expense of arable lands (Fig. 2E). As a result of large-scale agriculture, local population rapidly decreased. Due to privatisation of lands since the transition to market economy in the 1990s there has been an increase of

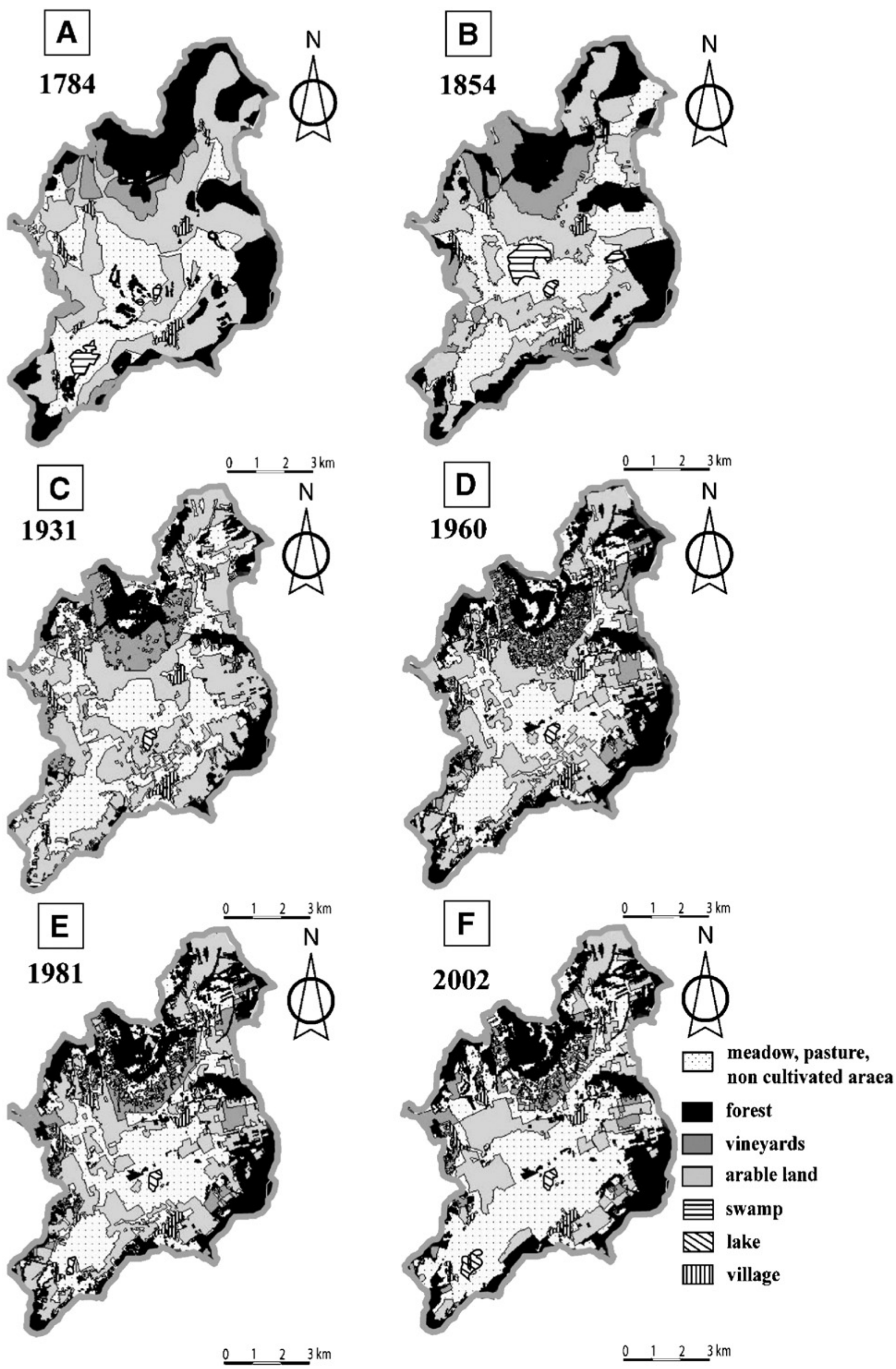

Fig. 2. A–F: Land use for the indicated years in the Kali Basin. See text for details.

uncultivated areas, particularly in areas of labour-intensive vineyards (Fig. 2F). In the lack of harmonised land management, land use pattern has become heterogeneous and spatially fragmented (Fig. 2F). New landowners are often temporary residents for touristic purposes, particularly in the scenic historic vine growing hillsides, thus vine growing and necessary erosion control have declined in the last decade.

3. Methods

3.1. Agricultural soil properties and evaluation, and erosion modelling

Soil map and data was produced according to the National Soil Mapping Standards (Stefanovits et al., 1970) and it is available only for the flat basin interior and thus

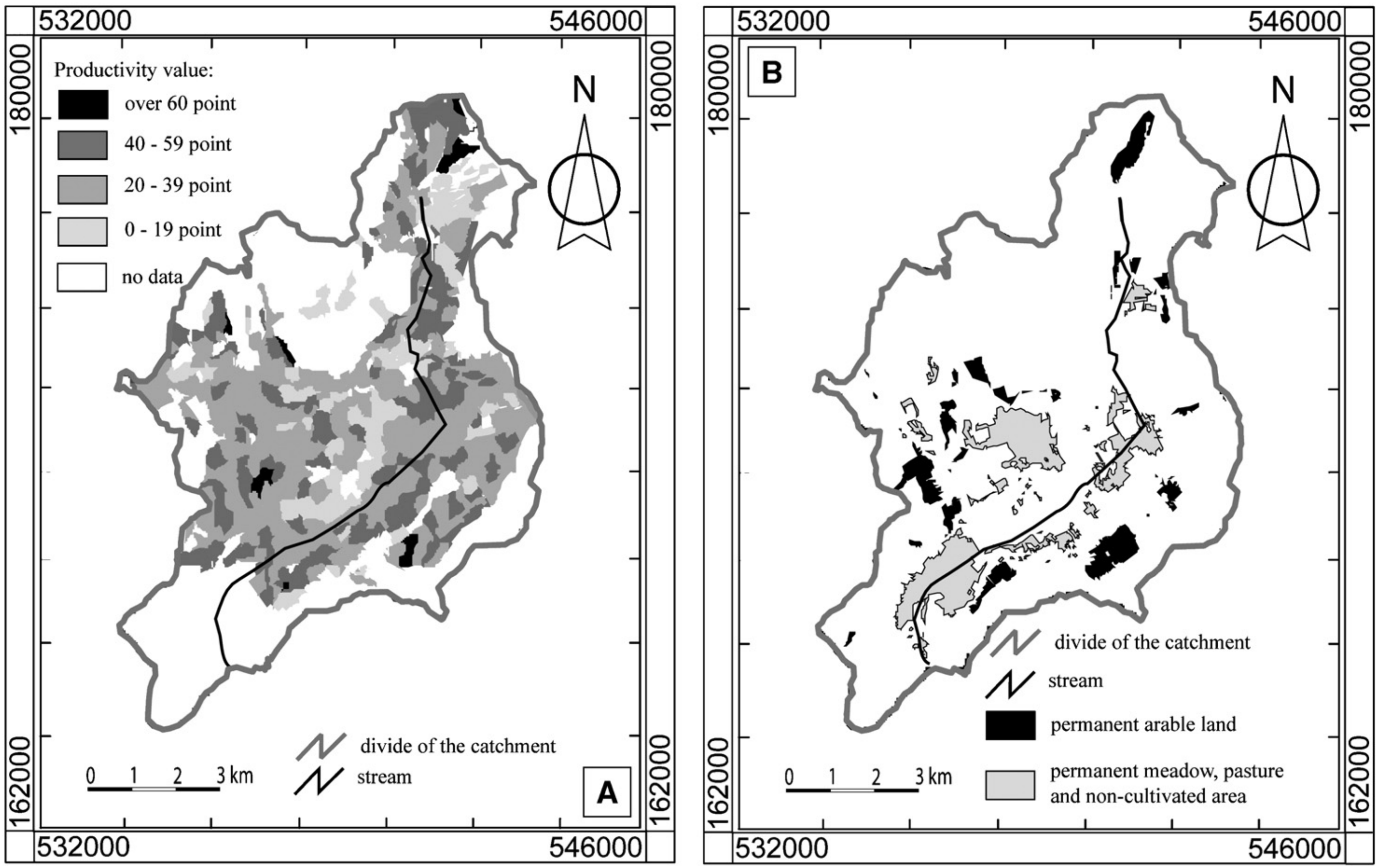

Fig. 3. A. Soil productivity values calculated for SMUs in the soil map. Note that the soil map covers only the flat basin interior. B. Spatial distribution of permanent arable and non-arable (the latter including meadow, pasture, non-cultivated areas in this study) lands. See text for details.

analysis is confined to this area in the present study (see Fig. 3A) (Szaszine, 2000). Among the 17 parameters based on field and laboratory measurements recorded for each soil mapping unit (SMU) polygon (Matene, 1990a), the following five parameters were selected assuming to be indicators of soil degradation in this area (Matene, 1990b): (1) Ph of the A horizon (recorded as pH intervals; see Figs. 5A, B), (2) depths of $CaCO_3$ accumulation (recorded as depth intervals in cm; see Fig. 5C, D), (3) $CaCO_3$ content in the accumulation zone (recorded as percent intervals; see Figs. 5E, F), (4) organic matter content in the A horizon (recorded as 'non-humic', 'weakly', 'intermediately', 'strongly' and 'very strongly' humic soil classes), and (5) erosion level (recorded as 'accumulation area', 'non-eroded', 'weakly', 'medium', 'strongly' and 'fully' eroded soil classes). According to the definitions, soil section is 'weakly', 'medium', 'strongly' or 'fully' eroded if <30%, 30–70%, >70% or 100% of the ideal complete section of a given soil type is eroded, respectively (Baranyai, 1989; Thyll, 1992). Note that in this way all these parameters are represented by classes defined by numerical intervals. The 'ideal complete section' is based on field observation in the mapped locality. The actually studied soil section is compared to a non-eroded, 'complete' section in the closest vicinity in the site. Thus, 'completeness' of section is site specific and it is based on the subjective judgement of the mapping expert. The broad percent intervals ensure consistency among individual judgements.

Unlike the first three parameters, organic matter content and erosion level are defined and measured relative to the ideal complete section for each soil type (Baranyai, 1989), thus these parameters already include the differences among the mapped soil types (compare Figs. 4 and 5).

The sixth soil parameter used in this study, the 'soil productivity value' was also derived from the map according to the National Soil Evaluation Criteria (Stefanovits et al., 1970). This method ranks soil productivity on a scale of 100 scores (Fig. 3A). A theoretical maximum 100 score productivity value is attached to *each soil type* and then various scores are subtracted from this based on soil degradation characteristics described by the mapped 17 parameters.

'Permanent arable land use' and 'permanent non-arable land use' (the latter including meadow, pasture and non-cultivated land uses in this study) areas shown in Fig. 3B were defined as those areas where these land use classes have remained the same for all the land use maps (see Fig. 2). Finally, this land use map (Fig. 3B) and the soil map with the above parameters were combined to study the impact of permanent agricultural land use on soil properties, assuming

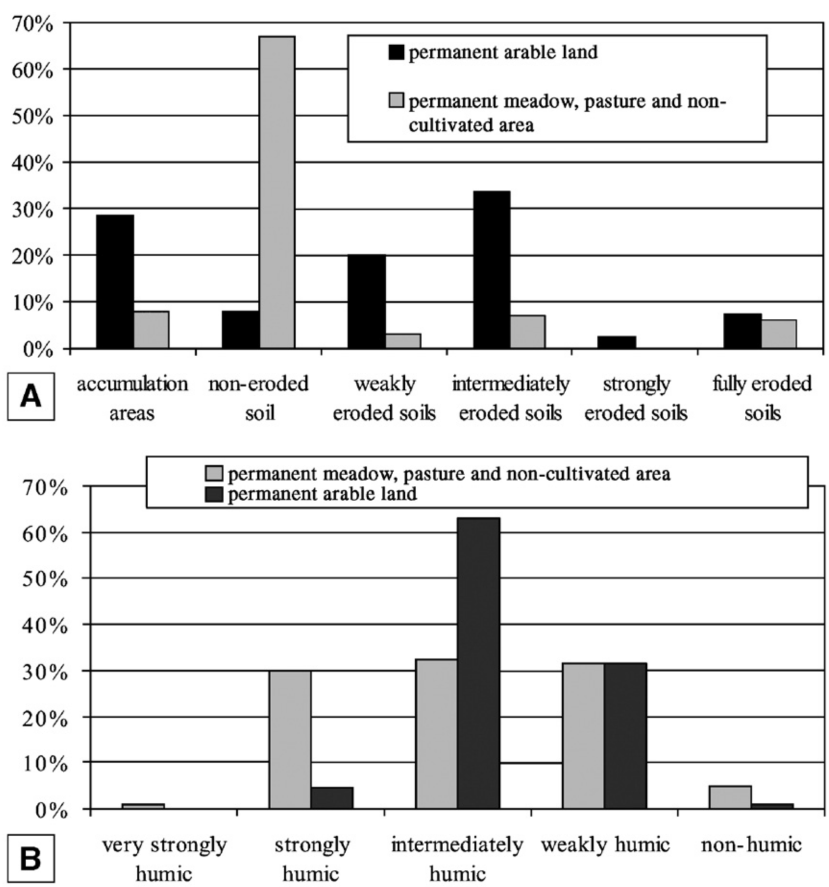

Fig. 4. A. Area proportion (%) of mapped soil erosion categories in studied permanent land use areas. B. Area proportion (%) of mapped humus content categories in studied permanent land use areas. (Compare to Fig. 3B.)

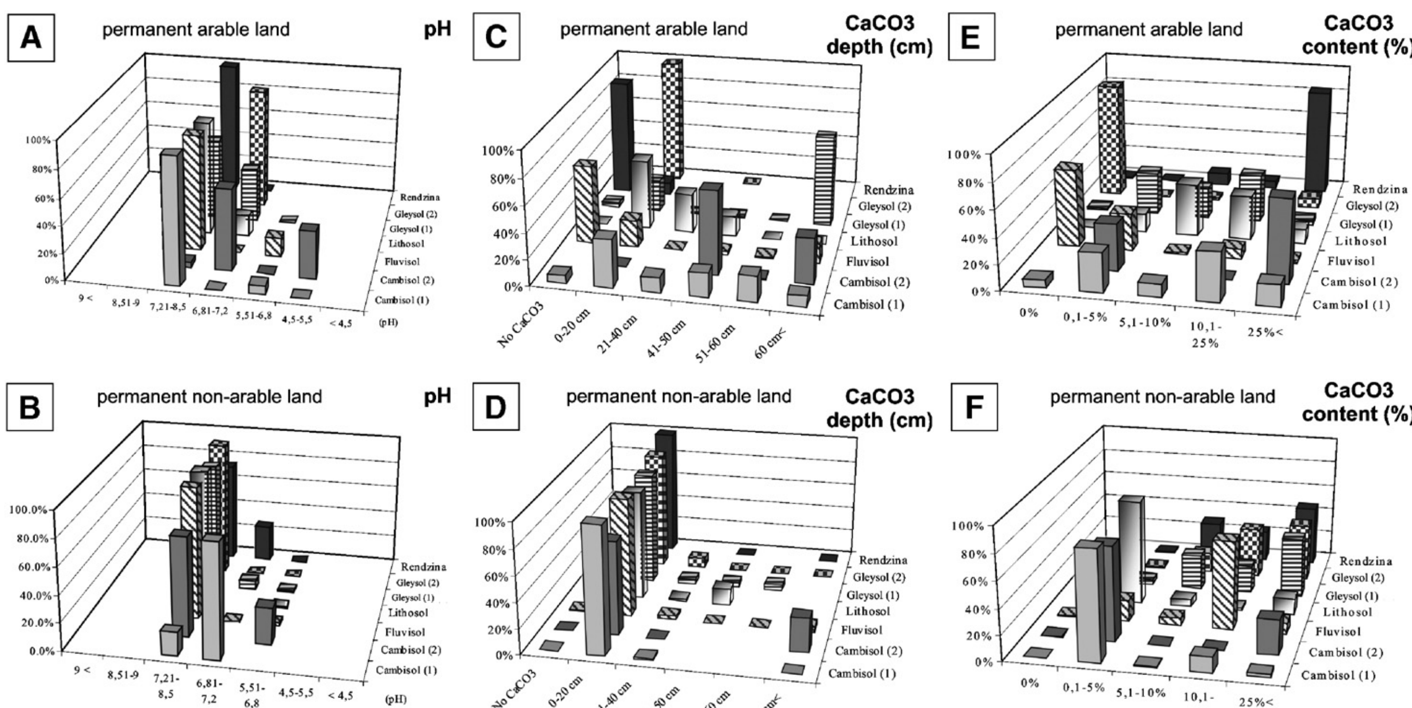

Fig. 5. A and B. Area proportion (%) of pH categories in topsoil of different soil types under permanent arable lands and non-arable lends, respectively. C and D. Area proportion (%) of carbonate accumulation depth categories in the accumulation zone of different soil types under permanent arable lands and non-arable lends, respectively. E and F. Area proportion (%) of carbonate content categories in the accumulation zone of different soil types under permanent arable lands and non-arable lands, respectively.

that long-term arable land use may lead to soil degradation. Thus, for the two 'permanent' land use classes (Fig. 3B) the area percent of all classes for each of the above six parameters was calculated and used for evaluation. Note that while land use change (or no change) was derived from historic land use map series, soil properties were derived from only one recent soil map.

Erosion and sediment transport modelling used the WATEM/SEDEM distributed erosion model (Van Oost et al., 2000; Van Rompaey et al., 2001) in this study. This model combines elementary process equations with a pixel-to-pixel routing topology to predict sediment fluxes, similar to the approach of Coulthard et al. (2002) and it uses suspension load data at the outlet of the catchment to calibrate the model. The model has three main components: (1) the assessment of a mean annual soil erosion rate for each grid cell, (2) the assessment of a mean annual transport capacity for each grid cell, and (3) a sediment routing algorithm that redistributes the produced sediment over the catchment taking into account the topology of the catchment and the spatial pattern of the transport capacity. The erosion component consists of an adapted version of the Revised Universal Soil Loss Equation (RUSLE; Renard et al., 1991) to calculate the mean annual soil erosion rate defined as:

$$A = R{\cdot}K{\cdot}L{\cdot}S{\cdot}C{\cdot}P \tag{1}$$

where A: mean annual soil erosion rate (kg m^{-2} y^{-1}), R: rainfall erosivity factor (MJ mm m^{-2} h^{-1} y^{-1}), K: the soil erodibility factor (kg h MJ^{-1} mm^{-1}), L: the slope length factor (–), S: the slope gradient factor (–), C: cover and management factor, P: supporting conservation practices.

For each grid cell a mean annual transport capacity was assessed according to Jordan et al. (2005) in the Kali Basin, where mean annual transport capacity is defined as the maximum mass of soil that can exit a grid cell per unit length of the down-slope face of the square (Desmet and Govers, 1995; Van Oost et al., 2000).

Finally, once the mean annual erosion rate and the mean annual transport capacity were known at each grid cell, the eroded sediment was routed down-slope to the river network along flowpaths from grid cell to cell. When a flowpath reached a river-cell all the sediment was delivered to the river. For each cell the amount of sediment input was added to the amount of soil erosion in that cell and the total sediment was routed further down-slope. If this sum exceeded the transport capacity then sediment output from the cell was limited to the transport capacity. The output of the model consists of (1) a pixel-map showing RUSLE local sediment production and (2) a pixel-map representing the amount of sediment erosion or deposition at each pixel. Furthermore, the amount of sediment that reaches the river channels was calculated giving the total catchment sediment yield (SY in Mg y^{-1}) that leaves the catchment at the outlet. Catchment specific sediment yield value (SSY in Mg h^{-1} y^{-1}) was calculated by dividing the total catchment sediment yield value by the catchment area.

After calibration and validation of the sediment flux model using calibration data from 1981–1989, erosion and sediment yield were assessed for land use scenarios in 1784, 1854, 1931, 1960 and 2002, according to Jordan et al. (2005). Since the objective of the present study was not the reconstruction of actual historic sediment fluxes but the analysis of land use effects, use of same rainfall erosivity, soil erodibility, topographic conditions and land management practices as for the calibrated 1981 scenario was justified. In fact, historic climatic data shows little variations in the study area and erosion model uncertainty is well within that of historic rainfall data (Racz, 1999). Also, topographic conditions and soil erodibility are not expected to change in the studied flat basin interior (Gabris et al., 2003) (Figs. 1A and 3A). In the lack of data, impact of land management practice changes cannot be assessed for the actual sediment fluxes. Thus, the calculated sediment fluxes might be similar to actual historic sediment fluxes, although their assessment was not the objective of this study.

3.2. Data preparation with GIS

Historic land use for years 1784 and 1854 was digitised from the 1:28,000 scale First Military Survey Map and Second Military Survey Map, respectively. The 1:25,000 scale Third Military Survey Map was the source for 1931 land uses, while the 1960, 1981 and 2002 1:10,000 scale National Topographic Maps were used for land uses in successive years (Fig. 2). The 1:10,000 scale soil map was digitised and soil properties available for SMU polygons were added to the attribute table of the map (Fig. 3A).

For erosion modelling, RUSLE C factors were derived from the digitised land use maps (Jordan et al., 2005). Soil texture data for the K factor in the erosion model was taken from the 1:10,000 National Agricultural Soil Map (Matene, 1990a) and, where soil data was absent, it was derived from the geological map (Jordan et al., 2005). Slope angle, flow direction and upslope drainage area was calculated from a 10 m resolution DEM (Jordan et al., 2005). The P factor was assumed to be constant and equal to 1 for the study area. Rainfall erosivity (R factor) values were assessed from monthly rainfall records using a formula proposed by Renard and Freimund (1994).

Population data registered for the villages were obtained from census records and they were compared to the size of land use types calculated for the agricultural area belonging to each settlement from the digital land use maps.

Digitisation, geo-referencing and database development was done with the ArcView® GIS software. Area and area proportions were calculated using ArcView® GIS procedures (ArcView, 2002). Identification of areas of permanent land use types (Fig. 3B) used simple GIS overlay procedures for the successive land use maps, while calculation of the digital soil productivity map (Fig. 3A) for the SMUs required the implementation of the arithmetic procedures of Stefanovits et al. (1970) on the soil parameters in the soil map attribute table.

4. Results and discussion

According to Fig. 4A, the area proportion of eroded soils under permanent arable lands is significantly higher than under permanent non-arable lands. On the other hand, proportion of non-eroded soils is close to 70% under areas that have been permanently non-arable lands during the last 200 years (Fig. 4A). Although erosion depends on many factors other than land use, including slope and slope length, the studied areas are all found in the flat basin interior thus these factors can have only subdued effect on erosion. It can be concluded that the observed higher erosion levels under arable lands as compared to non-arable lands is attributable to long-term cultivation. The higher accumulation area proportion under arable lands (Fig. 4A) is due to that farmers preferred these areas for cultivation due to deeper fertile soil profiles (Szilassi, 2003). Humus content under arable lands also tends to be lower than under non-arable lands (Fig. 4B) confirming that erosion and soil degradation is higher in cultivated lands in the Kali Basin. These results show for the first time that loss of humus due to arable land use is an actual and measurable process in this region (Szilassi, 2003).

Based on the analysis of chemical soil properties of soils under permanent arable and non-arable lands, the following observations can be made. Comparison of Fig. 5A and B shows that while soil pH is mostly in the range of 7.21–7.85 in permanent non-arable lands, soils tend to be more acidic (pH = 6.81–7.2) in arable lands for all the mapped soil types. Exceptions are only cambisol(1) and rendzina, however rendzina has generally higher carbonate content to buffer acidification, while explanation for cambisol(1) would require further detailed study. The observed acidification in arable lands is attributable to the application of fertilisers and increased leaching due to ploughing (Matene, 1990a). Effect of ploughing through increased leaching can be seen in Figs. 5C and D, too: carbonate accumulation zone tends to be at greater depths in all soil types under permanent arable lands. Exceptions are gleysol(2) and fluvisol, however proportion of carbonate free areas is very high for these soils according to Fig. 5C. Finally, when soil carbonate content in the carbonate accumulation zone is compared (Figs. 5E and F) it can be observed that carbonate content tends to be higher for permanent arable lands than for non-arable lands. This can be explained by the application of fertilisers and intensive leaching of upper soil horizons due to cultivation.

Jordan et al. (2005) investigated the effect of sediment transport controlling structures such as roads, streams and ponds in sediment delivery to the catchment outlet. That study concluded that the modelled sediment transport controlling structures have little impact on average sediment production and on total and relative catchment sediment loss in the Kali Basin (Figs. 6A, B and C).

Results of numerical erosion modelling show that average sediment production has been steadily decreasing in the studied period (Fig. 6A), while total sediment export from the basin increased (Fig. 6B). The catchment sediment delivery ratio curve in Fig. 6C shows a steady increase in the sediment delivery ratio indicating that the overall sediment retention capacity of the Kali Basin catchment has decreased. In other words, while average sediment production decreased (Fig. 6A), the absolute (Fig. 6B) and relative (Fig. 6C) amount of produced sediment leaving the catchment increased.

In order to explain these results of erosion modelling land use pattern was studied by (1) calculating for each land use scenario the area percent of each land use class in the total catchment area, and (2) by assessing landscape fragmentation as expressed by the number of land use polygons in each scenario (Jordan et al., 2005). Total area share of villages, orchards, marshlands and lakes has been insignificant with minor fluctuations only. The small values and minor fluctuations of total vineyard area (that have relatively high erosion potential) cannot be related to the observed sediment production trend (Fig. 6A), either. Relative area of forests also remained constant around 28%.

Meadows and pastures of low C factor however seem to steadily replace more erodable arable lands (Fig. 6D) that explains well the decreasing trend in sediment production in Fig. 6A. Increase in catchment sediment loss (Figs. 6B and C) is related to increase in sediment transport efficiency by change in spatial land use pattern. In order to account for differences due to various scales of original land use maps, all land use maps were resampled to the lowest 1:30,000 scale. Firstly, according to Fig. 6E, a steady and significant increase in landscape fragmentation is obvious for most of land use classes. Secondly, the first two ‘old’ scenarios, and the last three ‘modern’ scenarios seem to be similar among them, and the 1931 scenario is intermediate between these two, in general (Fig. 6E). Also, the sudden increase of land use polygons (about 2–10 times) before and after the 1931 scenario is apparent. These observed trends are consistent with the pattern observed for the catchment delivery ratio in Fig. 6C. The drop in landscape fragmentation for the 2002 scenario, as expressed by the total number of polygons in Fig. 6E, is due to the socio-economic transition in the last decade that was described earlier in this paper.

Results show that in spite of the significant decrease of arable lands resulting in decrease of average sediment production, suspended sediment loss through the drainage system has increased above the average level of ‘old’ (1784, 1854) land use scenarios. The modelled decrease in average sediment production (Fig. 6A) can be explained by the increase of meadow and pasture areas at the expense of arable lands (Fig. 6D) in general, while the absolute and relative increase in catchment sediment loss (Figs. 6B and C) can be explained by increase in landscape fragmentation

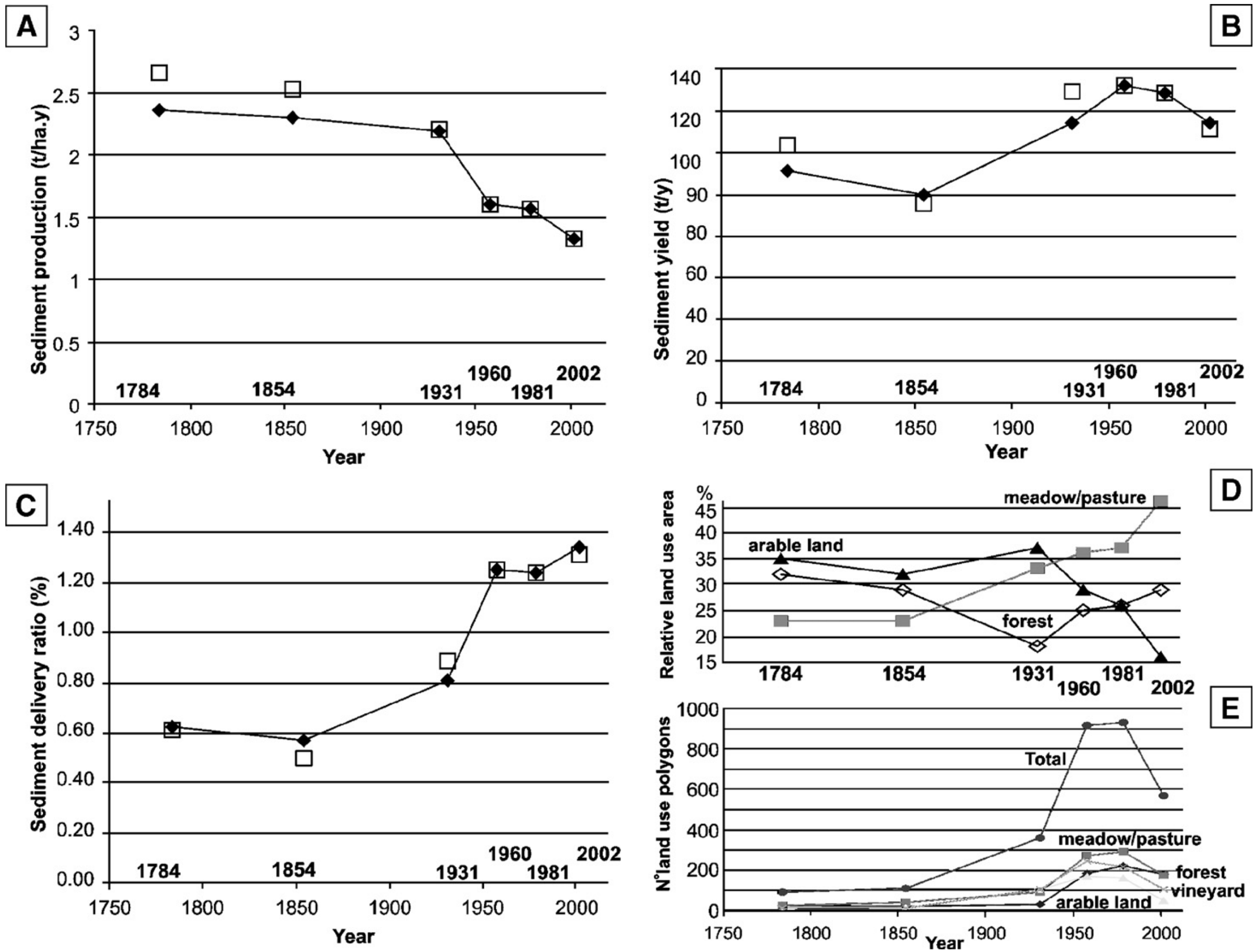

Fig. 6. Evolution of soil erosion in the Kali Basin. A. Average yearly sediment production (Mg h^{-1} y^{-1}). B. Average yearly sediment export (Mg y^{-1}). C. Catchment sediment delivery ratio (%). For A, B and C solid and empty rectangles show model runs with the 1981 scenario and actual stream and road networks, respectively. D and E. Change in area (% of total catchment area) and in number of polygons for selected land use types, respectively. Common time scale is shown in the *X* axis in E, and bold figures are years of the land use maps in D. See text for details.

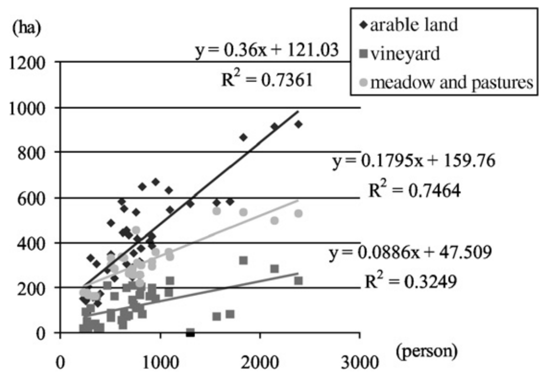

Fig. 7. Connection between the number of inhabitants and the size of arable and non-arable lands for the settlements of the Kali Basin, based on 43 samples. Regression for vineyards is based on 28 samples.

(Fig. 6E). Increasing landscape fragmentation opens corridors for sediment transport to the receiving drainage network. Formerly uniform land use patches blocking sediment transport along flow lines become fragmented hence increasing the possibility of sediment through-flow. Modelling results also allowed concluding that average sediment production does not exceed the 2 Mg h^{-1} y^{-1} in the modelled scenarios that, in general, is low and acceptable by national standards (Centeri and Csaszar, 2003).

Since self-sufficient agriculture was the most important and almost exclusive economic activity of the local inhabitants for centuries, a relationship between the size of land use types and population for each settlement can be assumed before the introduction of large-scale industrial agriculture in the 1960s. From this perspective, the most relevant land use types are (1) arable land for growing cereals for nourishing of the inhabitants, (2) meadows and pastures for animal husbandry, and (3) vine as the only agricultural product produced for market. According to Fig. 7, there is significant correlation between the population and total area of arable lands ($r = 0.86$), and total area of meadows and pastures ($r = 0.85$) for each settlement (see Fig. 1A), respectively. A significant but much lower correlation ($r = 0.57$) exists between population and total area of vineyards for each settlement. The weak correlation

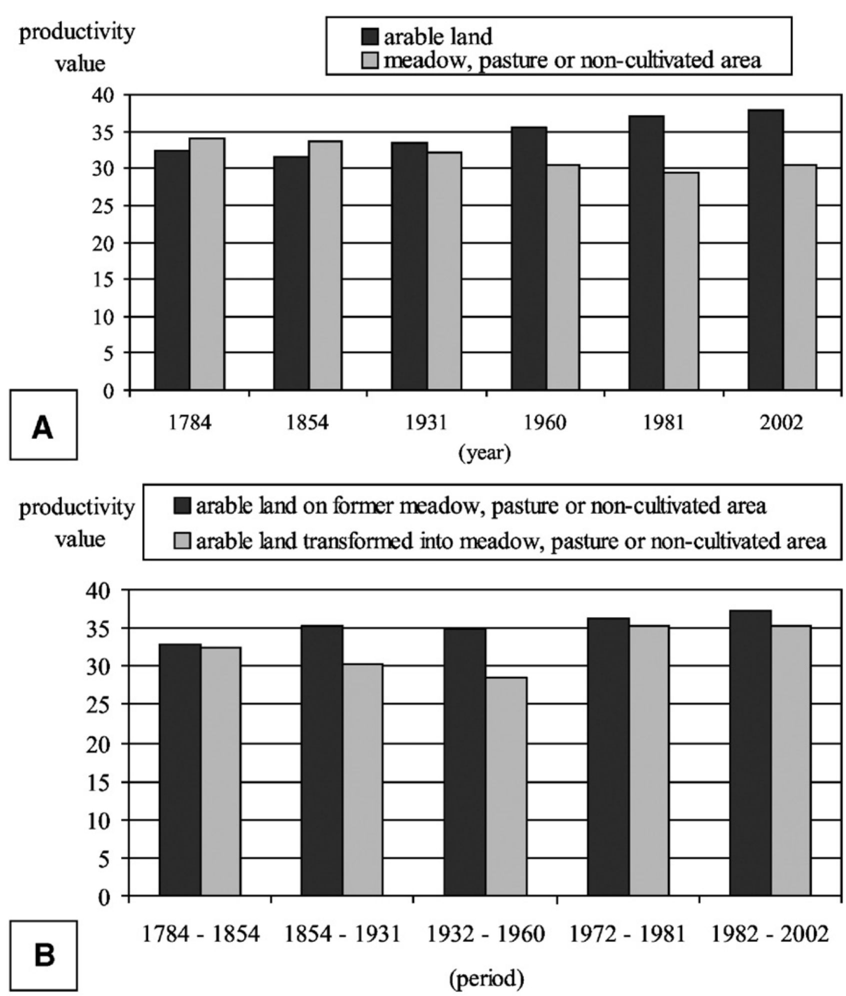

Fig. 8. A. Average soil productivity values for arable and non-arable lands for the indicated years (compare to Fig. 2). B. Average soil productivity values for areas with changing land use type between the two permanent land use types, between the successive land use map (see Fig. 2). See text for details.

for vineyards is explained (1) by the fact that vine was the only good produced for (mainly foreign) market, and (2) by the strong influence of local topography on vine growing (Szilassi, 2002), thus total area of vineyards depended on other factors then population, as well.

Relationship between soil quality, as expressed by the soil productivity value, and land use type was also studied. According to Fig. 8A, average soil productivity values for arable lands increase with time, while non-arable lands tend to occupy less and less valuable soils in the Kali Basin. Fig. 8B shows average soil productivity values calculated for two land use change types: (1) non-arable lands turning into arable lands, and the reverse, that is (2) arable lands turning into non-arable lands between the six successive historical land use maps (see Fig. 2). It can be seen that average soil productivity values have increased for the arable lands and they have decreased for the non-arable lands (Fig. 8). Also, the soil productivity values for areas that turned from non-cultivated to arable lands are higher than for those areas where arable land use was stopped (Szilassi, 2003). These lead to the conclusion that local farmers have learnt with time which are the areas of 'better' soils to cultivate as arable lands, and which are the areas with soils more suitable for meadow, pasture and non-cultivation land uses. This process was relatively slow because differences between soil productivities are little in the Kali Basin and it took time to recognise these differences in crop yield.

These results show that land use changes in the Kali Basin depended on natural and social factors. While (prior to the introduction of large-scale agriculture in the 1960s) the total *size* of arable lands was mostly affected by pressure of the population size, the *spatial pattern* of arable lands was affected by the productivity character of soils. It can be concluded that the study of the underlying mechanisms that trigger land use changes is important for long-term planning and sustainable management of soil resources in the Kali Basin (Szilassi, 2004).

5. Conclusions

Comparison of physical and chemical soil properties for areas used as arable and non-arable lands permanently during the last 200 years has shown that arable land use has lead to soil degradation in the Kali Basin. Soils in permanent arable lands are more eroded, they contain less humus and their upper layer is more acidic. A further impact of arable land use on soil properties is that $CaCO_3$ accumulation level is found deeper in these areas. The soils of arable lands have more $CaCO_3$ content in the carbonate accumulation zone than the soils of non-arable lands. Erosion and sediment transport modelling has shown that soil erosion and sediment production depend on the proportional area of land use types in Kali Basin, however, sediment loss from the catchment to Lake Balaton is related to landscape fragmentation (Fig. 6).

Land use changes in terms of size and pattern were influenced by different process in the Kali Basin triggered by natural and social factors. The total size of arable lands was mostly affected by pressure of the population size, the spatial pattern of arable lands was affected by the productivity character of soils.

The overall conclusion of this study is that besides the size and area proportion of land use types, land use pattern seems to be equally important in soil erosion and degradation processes, and for landscape development and future planning in the Kali Basin.

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