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Research article

How far back should we look? Long-term grassland changes to inform habitat restoration planning

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A B S T R A C T

Long-term time-series analyses are essential to avoid underestimating ecosystem loss and degradation and to prevent the shifting baseline syndrome when assessing habitat decline and setting restoration targets. Our aims were to (1) quantify historical area changes of semi-natural and transitional (degraded, regenerating, or intensive) grasslands in Hungary; (2) assess grassland continuity; and (3) evaluate grassland habitat loss across different historical reference points. For the analysis, we updated and supplemented the Habitat Trend Database of Hungary (5000 sample localities) with new field data collected in 2024, representing the past ten years of changes. The analysed database contains habitat data estimated for eight time points over the past 240 years. Our results show that grassland cover declined from 37.6% in the 18th century to 9.2% today, reaching its minimum extent (8.4%) in the 1980s. It is now increasing due to the expansion of transitional grasslands regenerating after the peak of cultivated area in the 1960s. Over the 240-year study period, the total loss of semi-natural grasslands accounted for 86.3%. 67% of Hungary's current semi-natural grasslands have remained continuous since the 18th century, while only 8.7% of the 18th-century semi-natural grasslands persist continuously to the present day. The average area of the studied Natura 2000 grassland habitats decreased by 38% since the 1980s, by 61% since the 1960s, and by 70% since the 1940s. We argue that if the original extent of habitat loss is not clearly documented and communicated, both the scientific community and society may gradually lose awareness of past ecosystem conditions.

1. Introduction

1.1. Importance of grasslands

Recent global biodiversity assessments identified alarming trends in species extinction and habitat loss (Fardila et al., 2017; Tilman et al., 2017), largely driven by human-induced transformation and deterioration of semi-natural habitats (Helm et al., 2006; Banks-Leite et al., 2020). Deforestation is acknowledged to be one of the most significant causes of biodiversity loss (Wiebe and Wilcove, 2025), while the consequences of the transformation and loss of semi-natural grasslands have received relatively little attention (Habel et al., 2013; Cousins et al., 2015; Buisson et al., 2022). Grasslands play a vital role in securing a sustainable future by providing a range of ecosystem services (Bengtsson et al., 2019; Schils et al., 2022), including substantial carbon storage (Veldman et al., 2015).

Grasslands with a long-term continuity (often referred to as old-growth, ancient, primary grasslands) are of special importance, as they have a number of characteristics that make them unique, such as structural and functional complexity, high species richness, dominance of long-lived perennials, and complex belowground structures (Cousins and Eriksson, 2002; Hejcman et al., 2013; Buisson et al., 2022). Certain semi-natural grasslands in Central Europe are particularly important for biodiversity, with several ranking among the world's most species-rich plant communities at small spatial scales (≤50 m^2; Wilson et al., 2012; Habel et al., 2013; Biurrun et al., 2021). According to a recent study, more than 80% of native vascular plant species have probably existed continuously since the Last Glacial Maximum, suggesting that these grasslands are much older than we might think (Molnár et al., 2023).

Natural disturbances (fire, large herbivore grazing, flooding) and long-term human activities (e.g., traditional grazing and mowing) are of primary importance in maintaining European grasslands since the Neolithic period (Hejcman et al., 2013; Habel et al., 2013). In addition, extensive human land use can result in high species richness even in grasslands that were developed more recently on historically forested or former agricultural land (Pärtel et al., 2007; Hejcman et al., 2013; Babai and Molnár, 2014).

1.2. Challenges of grassland categorisation

In European cultural landscapes, fine-scale mosaics of grasslands,

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forests, and cultivated land were typical over long periods, and these mosaics have been shaped by varying intensities of human activity (Hejcman et al., 2013). Their natural state, continuity, origin, age, and the extent of anthropogenic influences have a major impact on how we classify grasslands. This complexity structured grasslands along various gradients, leading to often overlapping (or even contradictory) terminologies and definitions of grassland types. In terms of origin, historical continuity, and ecological stability, the underlying concepts of natural, primary, or old-growth grasslands are similar, although the terminology differs across studies (Hejcman et al., 2013; Prach et al., 2017; Dengler et al., 2020; Nerlekar and Veldman, 2020; Veldman et al., 2015; Török et al., 2020; Buisson et al., 2022). One of the most controversial terms, however, is 'secondary grasslands' as their various interpretations refer to different sections of these gradient-like characteristics. According to one terminology, secondary grasslands are created by humans on the place of non-grassland vegetation (e.g., Dengler et al., 2020), a term that also includes species-rich near-natural grasslands, degraded grasslands, and fallow land. By the other terminology, secondary grasslands are degraded, early successional, regenerating, artificial, or intensive grasslands, or grassland that formed after the degradation of an old-growth grassland (c.f. Nerlekar and Veldman, 2020; Buisson et al., 2022; Veldman et al., 2015). These authors define the distinction between primary and secondary grasslands in terms of continuity and origin. Buisson et al. (2022) define grasslands formed after deforestation as derived-grasslands.

Many scholars in vegetation science define grassland habitats of higher habitat quality as semi-natural grasslands and categorise them into various habitat types or phytoconological categories. Diekmann et al. (2019) define semi-natural grasslands as "*unploughed mown or grazed areas with moderate fertilisation*". National and European habitat classification systems (Bölöni et al., 2011; Molnár et al., 2007; European Commission, 1992; Janssen et al., 2016; EUNIS HC, 2017) use the term 'semi-natural grasslands' to cover the entire range of natural and semi-natural grasslands in Europe.

In this paper, under the term 'semi-natural grasslands', we understand grassland areas that belong to an identifiable habitat category and, in general, represent higher-quality habitats. These grasslands can be extensively managed, mown or grazed. However, grasslands that cannot be classified under any type of 'semi-natural grassland' habitats are rarely included in these habitat classification systems (but see EUNIS HC, 2017). In our paper, we defined these types of highly degraded, intensively managed, young regenerating, or uncharacteristic grasslands, usually lacking specialist species and often dominated by disturbance-tolerant and weed species, collectively as 'transitional grasslands', and considered them to be distinct from semi-natural grasslands. Because of the ambiguous meaning, we avoided using the term 'secondary grasslands' in this study.

Neglecting transitional grasslands in habitat systems is one reason why we have limited information about the area and distribution of these grasslands at larger scales (but see Molnár et al., 2007, 2008; Biró et al., 2013; Ridding et al., 2020). Historical changes of these transitional grasslands are also rarely analysed, as long-term time-series analyses face methodological difficulties in this context (see the similar challenges of secondary forests reported in Biró et al., 2022). As a result, our knowledge of past trends in transitional grasslands is rather incomplete. In this study, we can rely on a unique database developed using the iterative information transfer method (Biró et al., 2018a, 2018b), which enabled us to assess historical changes both in semi-natural and transitional grasslands in Hungary over the past 240 years.

1.3. Long-term trends of grassland changes

Long-term time-series analyses are essential to avoid underestimating ecosystem loss and state when assessing declining habitat trends (Käyhkö and Skånes, 2006; Johansson et al., 2008). Without such a long-term perspective, awareness of past ecosystem conditions may gradually be lost, leading to the shifting baseline syndrome (McClenachan et al., 2012; Soga and Gaston, 2018). Habitat-level historical reconstructions at a century scale are rare in vegetation science, as they are often limited by inaccuracies and topographic incompatibilities of historical map sources, and the abiotic heterogeneity within definable land-cover patches (Biró et al., 2013). However, by integrating multiple historical sources, such as military and cadastral maps, archival travel diaries and botanical descriptions, with modern GIS and remote sensing, soil maps, long-term changes can be reconstructed or modelled in detail. This approach has been successfully applied to heathlands and moors (Lachat et al., 2022; Gimmi et al., 2011; Ridding et al., 2020), riverine landscapes (Hohensinner et al., 2013), forests (Wang et al., 2023; Biró et al., 2022) and to calcareous or acid grassland habitats (Poschlod et al., 2005; Hooftman and Bullock, 2012; Biró et al., 2013, 2018a; Ridding et al., 2020). These long-term historical reconstructions also enable identification of grassland continuity, which is closely linked to biodiversity and ecosystem functioning (Cousins and Eriksson, 2002; Johansson et al., 2008; Krause and Culmsee, 2013; Schmid et al., 2017).

1.4. Historical reference baselines in habitat restoration

Due to the continuous loss and deterioration of European habitats, the importance of reconstructed historical habitat changes is increasingly recognised in policy frameworks such as the EU Biodiversity Strategy for 2030 and the recently adopted Nature Restoration Regulation (European Commission, 2024). Estimating habitat-level changes over long time scales has been proposed as a means of defining threshold values for the endangerment status of habitats (since 1750; Bland et al., 2016). This approach is consistent with the Habitats Directive and the Nature Restoration Regulation, which adopt the concept of favourable reference areas (hereafter, FRA; European Commission, 1992; Hering et al., 2023).

In this context, FRA is defined as *"the total area of a habitat type in a given biogeographical or marine region at national level that is considered the minimum necessary to ensure the long-term viability of the habitat type and its typical species or typical species composition, and all the significant ecological variations of that habitat type in its natural range …"* (European Commission, 2024). Although the Nature Restoration Regulation introduced FRAs as baselines for restoring habitats to a favourable conservation status, it does not provide associated methodological guidelines (European Commission, 2024). Member States need to rely on the guidelines provided under the EU Birds and Habitats Directives in setting up FRAs (Bijlsma et al., 2019).

One commonly applied method for determining FRAs is the reference-based approach, which relies on estimates of the historical distribution or extent of a habitat in a period when it is assumed to have been in favourable condition (Bijlsma et al., 2019; Bonelli et al., 2021). These approaches typically incorporate factors such as the percentage of habitat loss and current trends. Other approaches emphasise the role of modelling methodologies, including habitat suitability and distribution modelling (Bijlsma et al., 2019). Historical reference baselines (a *quantitative snapshot* of habitat area, distribution, or quality at a selected point or period in the past) need to be established for the full range of habitat types addressed in national restoration planning to more accurately estimate FRAs (Bijlsma et al., 2019). In connection with this, the historical reference point is a *specific point in time* or a *year or period* used to define a historical baseline (Hilding-Rydevik et al., 2018). Since the selection of historical reference points for defining historical reference baselines remains under discussion in EU member states and among ecologists, a key contribution of our research is to analyse how outcomes depend on which historical reference points from the 20th century are applied.

1.5. Aims

Our objectives were to estimate 1) overall trajectories and historical area changes of semi-natural and transitional grasslands in Hungary, 2) to estimate the proportion of continuous grasslands over a 240-year period of time, and 3) to assess the grassland habitat loss across various historical reference points in order to inform the historical reference baselines for habitat restoration.

2. Methods

2.1. Study area

The spatial scale of the research was the whole territory of Hungary (93,030 km^2). In Hungary, the climate is subcontinental (mean annual temperature 9.5-11°C; annual precipitation 500-800 mm; Kocsis, 2018). The elevation ranges between 79 and 1014 m.a.s.l. (Kocsis, 2018). Hungary lies in Central Europe, and belongs to the Pannonian biogeographic region (Internet 1; Kocsis, 2018). The natural vegetation of the lowland regions consisted of *Quercus robur* dominated forest-steppes (natural sand, loess and alkaline forest-grassland mosaics), with wetlands in the lower elevations (e.g. reed beds, tall-sedge beds, tussock sedge communities, marshes, alkaline lakes, and fens). Alluvial softwood and hardwood forests were also characteristic of the river floodplains. The hilly-mountainous regions were dominated by *Quercus petraea, Carpinus betulus,* and *Fagus sylvatica* forests (Bölöni et al., 2011).

In the 18th century, human population density was relatively low, but it began to grow at an accelerated pace from the period of capitalisation in the 19th century (Jepsen et al., 2015). The communist transformation began after World War II, during which smallholder agriculture was reorganised into cooperative farming in 1961-1962. After the fall of communism in 1989, land was reprivatised (Jepsen et al., 2015). Common Agricultural Policy schemes had a substantial impact on land use since the accession to the European Union in 2004 (Mihók et al., 2017). The relatively strong nature conservation institution was established in response to the socialist nature-transformation efforts (Mihók et al., 2017). The proportion of protected areas by national law is now 9.2% of the country, and Natura 2000 areas, designated mostly in 2004, cover 21.39% of the country (Mihók et al., 2017; Internet 2).

2.2. Method

2.2.1. Use of field data for supplementing the countrywide habitat database

The *Habitat Trend Database of Hungary* (version 1.0; Biró et al., 2018b) is a countrywide database containing habitat data for 5000 randomly selected sample localities estimated in seven time steps (T1-T7; between the 1780s and 2010s). For the analysis, we supplemented this database with new field data on changes over the past 10 years (database version 2.0). The current, extended database now contains habitat data of 5000 randomly selected sample localities, estimated in eight time steps (T1-T8) over the past 240 years (1780s, 1850s, 1940s, 1960s, 1980s, 2000s, 2010s, and 2020s; Table 1). For database building and data management, ArcGIS 10.1 (ESRI), QGIS 3.10.1., and 3.16.7. (QGIS Development Team) softwares were used. Point localities of the database were randomly selected from the 267,813 grid cells of the *Hungarian Habitat Mapping Database* (Molnár et al., 2007). Besides main data sources (historical maps, satellite images and aerial photos), several additional historical and recent sources were used to build the *Habitat Trend Database of Hungary* (Biró et al., 2018b; for detailed methodology see Biró et al., 2018a).

Alongside information gathered from historical data sources, field data from the following main data sources were additionally used in the habitat estimation procedure: 1) Historical botanical field data from botanical diaries, publications, and archives; 2) field-based semi-natural habitat data of the *Hungarian Habitat Mapping Database* used for habitat interpretation of the 2000s; 3) targeted field surveys carried out in 2013-2014; and 4) repeated targeted field surveys of 296 point localities carried out in 2024.

Table 1
The studied eight time steps (T1-T8) over the past 240 years. Time periods are the decades of the time steps. Dates covered by data sources represent the broader span covered by the main sources (asterisks indicate cases when the date was summarised together with important additional sources of the date). Central dates represent the midpoint of each date covered by the data sources.

Time periods	Dates covered by data sources	Central dates	Main data sources
1780s	1780s	1783	First Military Survey, 1782–1785, HM HIM, Arcanum
1850s	1840s-1860s	1858	Second Military Survey, 1840–1866, HM HIM, Arcanum
1940s	1940s	1942	WWII Military Survey, 1940–1944, HM HIM, Arcanum
1960s	1950s-1960s*	1961	Corona satellite images, 1961-1969, USGS, RISP
1980s	1980s	1986	Landsat TM4–5, 1984–1987, USGS, RISP
2000s	2000-2005*	2002	Digital orthophoto series, 2000, 2005, BFKH
2010s	2010-2015*	2013	World Imagery satellite images, 2010–2013, ESRI
2020s	2023-2025	2024	Google Hybrid satellite images available under QGIS in 2024-2025; QGIS 3.10.1. and 3.16.7; QGIS Development Team Google Earth Pro satellite images, 2015–2025, 2025 Google LLC

Abbreviations: HM HIM: Hungarian Military History Museum, Budapest; Arcanum: Arcanum Adatbázis Kft (Arcanum Database Ltd.); USGS: U.S. Geological Survey EROS Center, Sioux Falls, South Dakota; RISP: Rectified by Interspect Kft; BFKH: Government Office of the Capital City Budapest, Budapest; ESRI: ArcGIS.10.1.ESRI/online-basemap-service/World-Imagery-Service.

2.2.2. Data interpretation - the iterative information transfer

For habitat estimation, we developed the iterative information transfer method, which enabled data enrichment through backward and upward information transfer between historical and recent sources (Biró et al., 2018a). The habitat estimation process consisted of three main stages: 1) identification of the exact point position; 2) estimation of land-cover types in the point locality in each time step; and 3) iterative habitat estimation procedure in the point localities that had grassland, wetland, or forest land-cover type at any studied time (see Biró et al., 2018a for further details).

In this procedure (third stage), selection of the potential habitat types was carried out based on land cover and abiotic features, such as geomorphological position (e.g. slope, exposure, microrelief), water supply, soil type, bedrock, and soil extremities (e.g., alkaline, sand, rocky, and wet soils). We iteratively selected potential habitat types, progressing through the abiotic features in a step-by-step manner. In each step, all main and additional sources relevant to the investigated feature were used to enable backward and upward information transfer between sources, thereby efficiently increasing the information available for habitat selection. Following this, we reduced the list of potential habitat types based on landscape and vegetation features, including biogeographical position, landscape context, land use, continuity of vegetation in the locality, and possible habitat transformations locally between periods (e.g., due to hydrological changes). In localities where semi-natural habitats were still present, we used actual and recent field data and determined probable past habitat types in a retrospective way. For this, we used additional sources available for the point locality (e.g. archival botanical descriptions, field surveys, and other thematic map sources (Biró et al., 2018a). To detect and trace the timing of possible past disturbances or temporary cultivations not visible in the map sources but recognised in the field survey, 20th- and 21st-century remote sensing sources were especially important. We used archival and recent aerial photography (Internet 3-4) as additional sources for this process, and high-resolution archival satellite imagery (Corona Spy satellite images, USGS, 1960s) as a main source (Table 1).

During the iterative infromation transfer, grassland, wetland and forest land-cover types were classified in each locality and each time step into national habitat categories with an identification of the most probable 2 habitat types with estimation probabilities indicated in the GIS database (for *Hungarian National Habitat Classification System* see Bölöni et al., 2011). Based on the estimated habitat types, an additional labelling category was also assigned to each locality at each time step. During this procedure, we labelled the localities with semi-natural habitat types, as *'semi-natural grassland'*, *'semi-natural forest'*, *'semi--natural wetland'*, *'transitional grassland'*, *'secondary forest'*, and *'secondary wetland'*. As these land-cover-level categories were derived from finer-scale habitat categories, we refer to these as '*habitat-based land-cover types*' (HBLC). HBLC types inlude *'cultivated land* 'and *'built-up & infrastucture'* areas.

2.2.3. Data analysis

For data exports, Python-scripts running in ArcGIS 10.1 ESRI were used. The output Microsoft Excel sheets contained the summarised habitat frequency and relative frequency data over time, and their visualisation on trend graphs (number of habitat localities weighted by the estimation probabilities). The required output groups of the summarisation were defined in an input Microsoft Excel sheet. For the analysis, we used seven time intervals (INT1–INT7) between the eight time steps (T1–T8) over the past 240 years.

We referred to natural and semi-natural grasslands as 'semi-natural grasslands' based on the terminology in the *Hungarian National Habitat Classification System* (Molnár et al., 2007; Bölöni et al., 2011; for definitions see also Török et al., 2020), which does not differentiate between natural and semi-natural grasslands (Molnár et al., 2007, 2008; Bölöni et al., 2011). We referred to uncharacteristic, degraded, regenerating, and intensive grasslands as 'transitional grasslands'. For the analysis, we further classified grasslands by origin, based on whether the grassland area was marked as grassland, forest, or wetland on 18th- and 19th-century maps (Table 2).

We considered 'long-term continuous grasslands' the grasslands that were recorded continuously as semi-natural grasslands from the 1780s up to the present day. These grasslands are, by definition, continuous semi-natural grasslands in our study. Because of the large number of grasslands with forest and wetland origins, we also analysed forest- and wetland-derived grasslands.

Table 2
Classification of grasslands by origin based on land-cover types depicted in historical map sources from the 18th–19th centuries. FMS: First Military Survey, 1782–1785; SMS: Second Military Survey, 1840–1866; SNG: Semi-natural grasslands.

Terminology used in this paper	Short name	Abbreviation	Land-cover in the 18th century map (FMS)	Land-cover in the 19th century map (SMS)
Semi-natural grasslands with long-term continuity	long-term continuous grasslands	SNG-LC	Grassland	Grassland
Semi-natural grasslands with forest origin	forest-derived grasslands	SNG-FD	Forest	Grassland
Semi-natural grasslands with wetland origin	wetland-derived grasslands	SNG-WD	Wetland	Grassland

To make the database records suitable for analysis, we entered the appropriate land-cover level categories for each habitat data record. We referred to these as '*habitat-based land-cover types*' (HBLC) and added for the data records in each locality and time steps as follows: 1) 'semi-natural grassland' - records with grassland habitat types (e.g. sand grasslands, alluvial meadows); 2) 'transitional grassland' - records not identifiable as semi-natural grassland habitat types (uncharacteristic, regenerating, degrading, intensive grasslands; see also Biró et al., 2022).

In Section 3.1, we estimated overall changes in semi-natural and transitional grasslands using summarised habitat-level data. For this analysis, we obtained the frequency of HBLC categories (e.g. semi-natural and transitional grasslands, cultivated land, built-up areas) by summarising habitat frequency data. The data and data visualisation were exported from ArcGIS ESRI to a Microsoft Excel datasheet using Python scripts.

For the grassland continuity analysis (section 3.2), we assessed grasslands that have remained semi-natural since the 18th century. We carried out quantitative trajectory analyses, a method for detecting the direction of change, identifying the origin of a habitat, and estimating long-term habitat continuity over time. The trajectory analysis is a spatial-temporal analytical framework that characterises sequences of land-cover, land-use, or habitat transformations across multiple time periods (Käyhkö and Skånes, 2006). In this study, grassland continuities were calculated from the number of point localities using upward trajectory analysis on grassland localities. For this, we filtered the semi-natural grasslands (SNG) localities in the database exported directly from the ArcGIS geoinformation database. For semi-natural grasslands continuous since the 18th century, we started filtering SNG localities in the 1780s (T1) and continued filtering them in series at the following dates, thus covering all periods (upward trajectories). For grasslands that were forests in the 1780s, we first filtered SNG localities in the 1850s (T2), then filtered forest localities in the 1780s (T1), and finally continuous SNG records in sequence covering all remaining periods from the 1940s (T3) (upward trajectories). We carried out the same procedure for SNG localities that were wetlands in the 1780s (T1).

In section 3.3, we exported data for assessing historical reference points from ArcGIS ESRI to a Microsoft Excel datasheet using a Python script that summarises changes in eight Natura 2000 grassland habitats, separately for each habitat. From this dataset, we used data from the mid-20th century to the present to assess changes since the 1940s, 1960s and 1980s. Although these periods span roughly a decade, this approach is consistent with the definition allowing a reference point to be a 'specific date, year, or period'. For calculating habitat loss since the historical reference points, the central dates (Table 1) of data sources were used (computed as the average of the map or image dates for Biró et al., 2018a).

These historical reference points represent major landscape transformation periods of the 20th century (c. f. Jepsen et al., 2015; Prokopová et al., 2018): the last years of private land ownership in the 1940s; the main socialist land transformation carried out in the 1960s; and the last years of socialist land-use in cooperative farming systems in the 1980s. Although the 2000s (the end of post-socialist land transformations and the years around the EU accession) were highly important for vegetation, we did not consider this period as a reference point, as we argue that such recent dates can substantially underestimate historical reference baselines. Habitat loss from the 1850s was used solely to visualise the degree of loss relative to a far-distant point of time. We argue that this date cannot be used as a reference date for many countries, partly because of major landscape changes that have taken place since then (c.f. Jepsen et al., 2015), and partly because of methodological limitations related to the use of historical maps (c.f. Lieskovský et al., 2018). For each habitat, we calculated habitat loss relative to different reference points, expressing the ratio of the current extent to its past extent as a percentage of loss.

3. Results

3.1. Long-term grassland changes in Hungary

We estimated that grasslands covered 37.6% of Hungary's total land area in the 18th century (T1; Fig. 1A). This figure decreased to a minimum extent of 8.4% by the 1980s (T5) and is approximately 9.2% today (T8). In 2024 (T8), the total grassland area consisted of 47% semi-natural grasslands (SNG) and 53% transitional grasslands (TG). In T1 (1780s), SNG covered approximately 35% of the country's territory, while in T8, this proportion was approximately 4.8% (Fig. 1A; Appendix S1). SNG showed a continuous decrease over the seven time intervals, with an average annual loss rate of 0.36%. Until the 1940s, this stayed around this rate (0.34 and 0.39 in INT1 and INT2), then accelerated to the highest rate of 0.6 in INT3, and 0.43 in INT4. Later, the loss rates of semi-natural grasslands slowed down to 0.24 in INT5, and were recently below 0.1 in INT6-INT7.

Transitional grasslands increased from the 1960s onwards and now cover approximately 4.4% of the country (T8). The proportion of cultivated land was 26.5% in the 18th century, peaked at 63.6% in the 1960s (T4, the period of 'maximum cultivation'), and currently stands at 53% (Fig. 1A; Appendix S1). The built-up (incl. infrastucture) area increased fivefold over the 240-year study period (from 1.7% to 8.5%; Appendix S1). The total loss of SNG was 86.3% over the 240-year study period (including long-term continuous grasslands and grasslands of forest or wetland origin).

3.2. Estimating continuous grasslands in Hungary

Grasslands with long continuity showed different trends than grasslands with forest or wetland origin. About 67% of all current semi-natural grasslands have been continuous since the 1780s (100% = 240 SNG localities in T8) (Table 3). Additional 5.8% and 1.7% are continuous from the 1850s, being semi-natural forest and wetland, respectively, in the 18th century. 8.7% of the 18th-century semi-natural grasslands remained continuous until this day (100% = 1833 SNG locality in T1). Grasslands originating from forests and surviving as continuous semi-natural grasslands from the 19th century consisted of about nine percent of their 19th-century area. Similarly, grasslands originating from wetlands and continuous from the 19th century accounted for 5.2% of their 19th-century area.

Table 3
Results of the continuity analysis. Values of three types of continuity are shown. Percentage of semi-natural grasslands (SNG) compared to the semi-natural grasslands in the 1780s and at the present.

Percentage of continuous semi-natural grasslands in the case of three different continuity types	% relative to SNG in the 1780s	% relative to SNG at the present
Long-term continuous semi-natural grasslands (continuous since the 1780s)	8.7	66.7
Semi-natural grasslands with forest origin (continuous since the 1850s)	9.1	5.8
Semi-natural grasslands with wetland origin (continuous since the 1850s)	5.2	1.67
Recent semi-natural grasslands (appearing only from the 1940s)	0	25.8
100% (Number of localities)	1833 SNG localities	240 SNG localities

We found that 82.5% of the 19th-century grasslands were grasslands in the 18th century, 11.5% were forests, and 6% were wetlands. Grasslands with long-term continuity had been mainly Pannonian salt steppes and salt marshes (1530*; 42%) and Alluvial meadows (6440; 35%) in the 18th century, and the proportion of both *Molinia* meadows (6410) and Sub-pannonic steppic grasslands (6240*) were 6% (Fig. 1B).

3.3. Habitat loss considering various historical reference points

We compared the loss of Hungary's eight most important Natura 2000 grassland habitats, accounting for various historical reference points in the 20th century (Fig. 2). The eight studied Natura 2000 grassland habitats decreased by an average of 38% since the 1980s, and 61% since the 1960s. The decline since the 1940s was 70%. By comparison, since a far-distant point in time, the 1850s, this ratio was 81%. The annual loss rates since the 1980s and the 1960s are close to 1% (0.99%/year and 0.97%/year, respectively). The loss rate since the 1940s was 0.86%/year. By comparison, since the far-distant 1850s, the rate would represent a 0.49% annual loss.

Since the 1980s, two habitats (*Molinia* meadows, 6410, and Alluvial meadows, 6440) have declined with a loss of extent (61 and 72%, respectively) considerably higher than average, while three habitats, Sub-Pannonian steppic grasslands (6240*), Pannonian sand steppes (6260*) and Pannonian salt steppes (1530*), have declined below

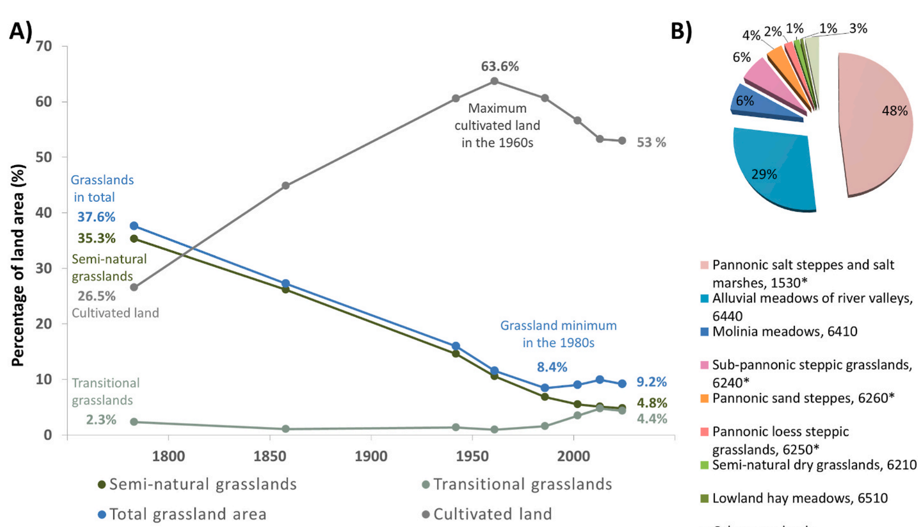

Fig. 1. A) Changes of semi-natural grasslands over the last 240 years in Hungary. Transitional (disturbed, regenerating, and intensive) and overall grassland loss are also shown. The overall grassland area reached its minimum extent of 8.4% in the 1980s. Cultivated land includes recently abandoned cultivated land. B) Proportion of habitat types in the 18th century in sample sites that have remained continuous grasslands since then. The numbers indicate the Natura 2000 codes for the habitats.

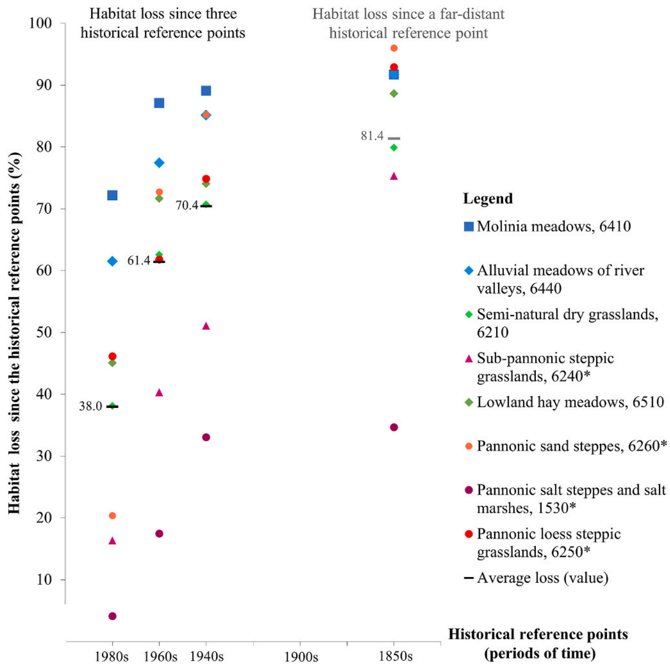

Fig. 2. The loss of Natura (2000) grassland habitats in Hungary since three different historical reference points in the 20th century (the 1940s, 1960s, and 1980s). We also estimated habitat loss since the 1850s, which allowed us to compare the data with a far-distant historical reference point. The numbers after the habitat names indicate the Natura 2000 codes. For calculating habitat loss since the historical reference points, central dates of data sources were used (Table 1). Average numbers show average habitat loss since the reference points. For data, see Appendix S2.

average (4-21%). Since the 1960s and 1940s, only two habitats, the Sub-Pannonian steppic grasslands and the Pannonian salt steppes, have been lost at a level below the average habitat loss. The loss of the Pannonian sand steppes was considerably above average (approx. 73% and 85% since the 1960s and 1940s, respectively). The *Molinia* meadows showed the greatest loss since the 1960s and 1940s (87 and 89%, respectively), but the decline of Alluvial meadows since these two reference points was only slightly less (77 and 85%, respectively).

4. Discussion

4.1. Long-term grassland changes in Hungary

Our results show that although the total grassland area has increased recently, the extent of semi-natural grasslands is currently smaller than ever before, at only 13.7% of its original extent in the late 18th century. In case of semi-natural grasslands, the total loss was 86.3% over the 240-year study period. Previous long-term research has indicated that the last two centuries brought significant changes in the extent of grasslands in other countries as well (Poschlod et al., 2005; Hooftman and Bullock, 2012; Ridding et al., 2020; Wang et al., 2023). For example, in south-eastern Sweden, semi-natural grasslands decreased by over 96%, largely due to afforestation (Cousins et al., 2015). In the Swedish island of Öland, 86% of the landscape was grassland in the early 18th century, and this was reduced to 8.7% by the 1990s, primarily replaced by arable land and forest (Johansson et al., 2008). In the United States, a similar decline was estimated for the period of 1800-1950, followed by a partial recovery from 1950 to 2000 (Wang et al., 2023).

The first major wave of agricultural intensification in the 18th-19th centuries mainly affected fertile soils, which were more likely to be converted into arable fields across the Northern Hemisphere (Jepsen et al., 2015; Skokanová et al., 2016; Hejcman et al., 2013; Wang et al., 2023). In Hungary, the highest speed of grassland loss occurred during the socialist period between the 1940s and 1960s. In the 20th century, the main land-change processes in Europe were cropland/-grassland dynamics, afforestation, deforestation, and urbanisation (Fuchs et al., 2015). In post-socialist countries, this second wave of agricultural intensification was closely linked to state-led collectivisation and 'socialist nature transformation' (Jepsen et al., 2015; Borvendég and Palasik, 2016; Skokanová et al., 2016; Prokopová et al., 2018). Areas previously unsuitable for cultivation, because of their low agricultural potential (e.g., sand dunes, salt steppes) became targets of 'socialist nature transformation' during this period (Borvendég and Palasik, 2016; Skokanová et al., 2016). In Hungary, increased mechanisation and the availability of new agricultural techniques made it possible to convert grasslands with poor soils to agricultural land: primarily vineyards and orchards on moving sand dunes, as well as rice fields in regions with saline soil (Biró et al., 2013; Borvendég and Palasik, 2016). Another consequence of agricultural intensification was that tractors and machinery replaced draught animals, significantly reducing the demand for hay. As a result, many hay meadows were no longer maintained through regular mowing and were later converted or abandoned (see also Biró et al., 2024 in Romania).

Our results showed that the grassland area continued to decline through the 1960s-1980s, reaching its minimum extent in the 1980s. This further decline, despite widespread abandonment of cultivation, was driven by ongoing afforestation and conversion to arable land of the remaining semi-natural grassland areas (Borvendég and Palasik, 2016; Skokanová et al., 2016; Biró et al., 2022). Similarly, a high grassland-to-cropland transformation was observed between the 1940s and 1960s also at the European level, followed by a slightly lower rate in the 1960s and 1970s (Fuchs et al., 2015). Semi-natural grassland declined sharply in the mid-20th century in the Dorset region (UK), and continued to decline at a slower rate thereafter (Ridding et al., 2020).

The minimum extent of grassland areas is rarely documented (but for forest minimum, see Barbier et al., 2010; Biró et al., 2022). In the Carpathian region, long-term changes show a minimum of forest area in the mid-20th century (Lieskovský et al., 2018; Munteanu et al., 2015). In Hungary, forest cover reached its lowest level in the first half of the 20th century (13% of the territory; Biró et al., 2022), whereas grasslands reached their minimum several decades later. In Hungary, the minimum extent of grasslands occurred decades after the maximum extent of arable land, similarly to several other European countries (c.f. Skokanová et al., 2016; Prokopová et al., 2018). In post-socialist countries, arable land generally reached its greatest extent during World War II or at the beginning of the subsequent socialist period (Hejcman et al., 2013).

The grassland area started to increase due to grassland regeneration following a peak in cultivated area in the 1960s. At the same time, semi-natural grasslands continued to decline after the end of communism, with abandonment increasing more rapidly than this decline. Furthermore, after the collapse of communism, the slowing decline in semi-natural grasslands can be attributed, in part, to the expanding role of nature conservation in Hungary (Mihók et al., 2017; Biró et al., 2018a). Post-abandonment grassland regeneration is a widespread phenomenon across Europe, from the Mediterranean to Scandinavia (Lasanta et al., 2017; Fayet et al., 2022). In Hungary, regenerating transitional grasslands have increased from the 1960s onwards and now cover an extent comparable to that of semi-natural grasslands.

This trend of cropland-to-grassland conversion is also observable across the EU between the 1970s and the 2000s (Fuchs et al., 2015). The map by Pazúr et al. (2024) shows a similar picture, indicating that during this period, Hungary's grassland areas mainly showed an upward trajectory and stability. The general trend in Central Europe after the 1960s was a slow regeneration of grasslands on abandoned agricultural land (arable fields, rice fields, vineyards, and orchards) (Prokopová et al., 2018; Skokanová et al., 2016; Lasanta et al., 2017). Where the landscape was diverse enough, and there was a sufficient supply of propagules, regeneration was significant (Hejcman et al., 2013; Teleki et al., 2020; Bartha et al., 2025; Halassy et al., 2019).

4.2. Grassland continuity

Grassland continuity can be expressed in relation to the current grassland area or the area of grasslands in a distinct period of the past, but such data are scarce and are available only for relatively small areas in Europe. For example, on a Swedish island, 62% of the existing grasslands have remained continuous since the early 18th century (Johansson et al., 2008). We found a similar percentage for a much larger area (about 67% of Hungary's current grasslands have remained continuous since the 18th century). This continuity was primarily associated with specific habitat types. It is remarkable that three-quarters of these long-term continuous grasslands originally consisted of Pannonian salt steppes and salt marshes (1530*), and Alluvial meadows (6440) in the 18th century. The latter, regularly flooded habitats, have remained grasslands to this day, as they were unsuitable for agricultural cultivation (Biró, 2009; Mihók et al., 2017). The cultivation of salt grasslands was also economically inefficient due to the soil's extremely high mineral content.

Detecting short-term cultivation (1-2 years) represents one of the main challenges in estimating the area of continuous grasslands, as such disturbances are rarely identifiable from available map series. Therefore, field surveys are particularly important for recognising this so-called land-use legacy in the present grassland and tracing back the time of the cultivation in a retrospective manner, using mainly 20th- and 21st-century archival and recent remote sensing sources (Internet 3-4). Short-term cultivation allows grasslands to regenerate well on abandoned fields, but the species composition might indicate that it was ploughed, even if only for a short period. Biró et al. (2024) assessed and classified land-use legacies and observed that, in many cases, these grassland changes can remain visible for decades, but in different ways, and may not be considered the same type of land-use legacy as cultivation lasting for decades or more.Despite the overall loss of grasslands during the late 18th and early 19th centuries, new grasslands also emerged (in connection with the European population explosion), particularly through forest clearance for haymaking and grazing (Hejcman et al., 2013). These long-established, forest-derived grasslands, especially the hay meadows, are of great importance due to their high species richness and diverse ecological roles (Hejcman et al., 2013; Cousins et al., 2015; Babai and Molnár, 2014). Some authors emphasise that it is misleading to view and handle these grasslands (often inheriting a large number of specialist species from the preceding forest-wetland mosaic) in the same category as degraded, recently formed, or post-disturbance ecosystems and refer to both of them as ‘secondary grasslands’ (Cousins et al., 2015; Veldman et al., 2015; Nerlekar and Veldman, 2020). We argue that the ambiguity surrounding the term ’secondary grassland’ reflects a deeper conceptual difference between classifications based on origin and those based on the habitat's ecological state (naturalness, level of degradation).

4.3. Historical reference points in habitat restoration

Our results highlight the value of long-term data and historical sources for identifying historical reference baselines, which serve as benchmarks for determining favourable reference areas (FRAs) in habitat restoration planning. We found that different historical reference points can substantially influence national estimates of habitat loss and, consequently, the historical baselines used in habitat restoration planning. Identifying periods of major anthropogenic impact can help guide the selection of appropriate reference points (Bull et al., 2014; Mihoub et al., 2017). In Central Europe, the 1940s (World War II) and the 1960s (socialist landscape transformation) are particularly relevant for vegetation change, while the 1980s reflect land-use conditions shortly before the collapse of communist regimes (Jepsen et al., 2015; Bürgi et al., 2017; Biró et al., 2018a; Prokopová et al., 2018). However, in practice, the limited availability of historical sources remains a major obstacle in setting historical reference points in many European countries (c.f. Mihoub et al., 2017).

Consequently, relatively few national-level historical analyses exist for European countries, and most of these rely on land-cover data rather than habitat loss estimates. Substantial differences may arise between land-cover-based and habitat-specific analyses, as the latter allow a more precise reconstruction of ecological processes (Biró et al., 2013). Even where habitat loss data are available, they often compare only two points in time rather than providing a longer time series (c.f. a single point in the past compared to the present), limiting the possibility of exploring variation across alternative historical reference points. Ideally, reference periods should extend beyond the last 20–30 years, but not too far back into the past, presumably with reference to the second half of the 20th century as a practical compromise. Our long-term dataset is illustrative in this respect, as national-level loss data for habitats listed in Annex I are available for six time points since the 1940s.

Our results using the 1980s, 1960s, and 1940s as reference points show that the choice of historical reference point leads to substantial differences in estimated habitat loss. Over the past 40–60 years, average habitat loss rates were approximately twice those observed over the last 160 years for the eight studied grassland habitats (see also Biró et al. (2018a); Three Natura 2000 habitat types in Hungary have experienced particularly large declines since the 1940s, with losses exceeding 85% of their area: Pannonic sand steppes (6260*), *Molinia* meadows (6410), and Alluvial meadows (6440). In contrast, when a more recent reference point is used (e.g., the 1980s), the estimated losses for the same habitats are substantially lower, at approximately 20%, 72% and 61%, respectively (see Appendix 2). Similar alarming figures of habitat loss have also been documented elsewhere in Europe. For example, alvar grasslands in Estonia lost approximately 94% of their area since 1930 (Helm et al., 2006). In Switzerland, moors lost more than 80% of their area and dry grasslands and meadows more than 95% since 1900 (Lachat et al., 2022). In the Stuttgart region of Germany, semi-natural calcareous grasslands declined by 61% between 1900 and 1990 (Poschlod et al., 2005).

Several European countries have already developed approaches for estimating favourable reference areas, which implicitly rely on particular historical reference points (van Eldik et al., 2024). Bijlsma et al. (2019) argue that, for the Netherlands, 1950 should serve as the reference point, which is also the year used for the European Red List of habitats. In Belgium (Flanders), historical reference calculations likewise trace back to the 1950 reference point, indicating a 78.8% decline in wetlands since that time (Decleer et al., 2016). Austria has assessed FRAs for approximately 80–85% of habitats using transparent and traceable methodologies that refer to the national habitat distribution data of 1995 as a historical reference point (Ellmauer, 2004). For example, in the case of *Molinia* meadows (6410), the FRA corresponds to 18.7–19.4% of the historical distribution, implying an 80.6–81.2% habitat loss since 1995. For comparison, the estimated loss of *Molinia* meadows since the 1980s is approximately 72% (see the same value above) in Hungary, suggesting somewhat less pronounced decline than that reported for Austria.

Overall, our results demonstrate how sensitive habitat loss assessments are to the chosen historical reference points. These findings highlight the importance of carefully selecting ecologically meaningful reference points in order to avoid underestimating long-term habitat decline in restoration planning. Comparisons across Europe suggest that historical trajectories have varied considerably depending on regional socio-political and environmental contexts. The variability in both the historical reference points applied by different countries and the resulting habitat loss estimates underscores the need for greater transparency and ecological justification when defining FRAs.

4.4. Implications for European restoration planning

Under the Habitats Directive, a favourable reference area for a given habitat type should, at a minimum, correspond to its distribution at the time the Habitats Directive entered into force (European Commission, 1992; Bijlsma et al., 2019). For countries that joined the EU after the Directive entered into force, the reference point could be understood as the year of accession (e.g., 2004 for Eastern European countries). Because the accession occurred after much of the major habitat loss in the 20th century, these recent baselines can substantially underestimate the historical losses and thus the corresponding FRAs. In the case of Hungary, the average loss of key Natura 2000 grassland habitats would not exceed 8%, and, due to habitat regeneration processes over the last two decades, several habitats would have values close to zero or even negative (Appendix 2; c.f. Internet 5). Regeneration had already begun in many areas of abandoned agricultural land, which can further reduce the apparent extent of historical loss if assessed only for recent conditions.

When relying solely on land cover data, it is important to note that this may obscure the ecological value of semi-natural compared to transitional grasslands, potentially leading to an underestimation of biodiversity loss. Consequently, there is a risk of falling into a shifting baseline syndrome (McClenachan et al., 2012; Soga and Gaston, 2018) if historical reference points are not properly chosen to capture the extent of past habitat loss. Restoration targets should therefore be determined using available historical data, ensuring that estimated historical baselines are ecologically meaningful and relevant to restoration planning.

We highlight that long-term continuous grasslands should receive greater conservation attention, as habitat continuity is essential for sustaining biodiversity and ecosystem functioning (Cousins and Eriksson, 2002; Krause and Culmsee, 2013; Schmid et al., 2017; Kimberley et al., 2020; Scherreiks et al., 2022). These grasslands should be prioritised in both restoration and conservation planning (c.f. Johansson et al., 2008; Hejcman et al., 2013; Schils et al., 2022). Our findings underscore the need to incorporate long-term historical perspectives into European restoration planning, particularly in post-socialist countries, where recent baseline data may underestimate the magnitude of long-term habitat loss.

5. Conclusion

Our assessment of overall change in grassland areas showed that over the past 240 years, semi-natural grasslands have declined and transitional grasslands have increased, representing opposing trends. The grassland area reached its minimum extent in the 1980s, and it is now increasing due to grassland regeneration following a peak in cultivated area in the 1960s. However, it is extremely important to distinguish between semi-natural and transitional grasslands, as interpreting grassland change solely through land-cover change can be misleading from an ecological and conservation perspective. As the choice of historical reference points for national restoration plans remains under discussion, the significance of our research lies in illustrating the different consequences of various reference points. We argue that if the original extent of habitat loss is not clearly documented and communicated, both the scientific community and society may gradually lose awareness of past ecosystem conditions. Consequently, there is a risk of falling into the shifting baseline syndrome if the periods used as historical reference dates for restoration are too short. Long-term historical datasets are therefore essential for defining meaningful restoration baselines.

Declaration of generative AI and AI-assisted technologies in the manuscript preparation process

During the preparation of this work, the authors used SciSpace and Research Rabbit for literature searches and Grammarly to check the language. After using these tools, the authors reviewed and edited the content as needed and take full responsibility for the publication's content.

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CRediT authorship contribution statement

Marianna Biró: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Writing – original draft. **Melinda Halassy:** Conceptualization, Writing – original draft. **János Bölöni:** Data curation, Formal analysis, Methodology, Writing – review & editing. **László Demeter:** Conceptualization, Writing – review & editing. **Zsolt Molnár:** Conceptualization, Methodology, Writing – original draft, Writing – review & editing.

Declaration of competing interest

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

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2026.130511.

Data availability

Data will be made available on request.

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