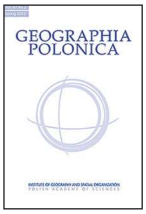

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FOREST ENCROACHMENT ON TEMPERATE MOUNTAIN MEADOWS – SCALE, DRIVERS, AND CURRENT RESEARCH DIRECTIONS

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Abstract

Meadows are characteristic features of the forested mountain landscape in the northern temperate zone. In terms of origin, they can be classified as natural, semi-natural and improved. Mountain meadows have great ecological value due to vast biodiversity and the ecosystem services they provide. However, over the past few decades, a significant decrease in their area has been observed in many places across the world. The purpose of this paper is to provide an overview of the scale and the main drivers of forest encroachment on temperate mountain meadows as well as to describe current research directions and methods. The observed decline in meadow area may be driven by natural factors related generally to climate change or may result from changes in land use. This process is investigated on a variety of spatial scales ranging from experimental plots to entire geographic regions. Studies on forest encroachment on mountain meadows are now carried out by researchers from many different countries. Nevertheless, there still does not exist a complex, multidisciplinary approach and comparative studies for different mountain ranges are not found in the literature.

Key words

origin of meadows • forest encroachment • secondary succession • climate change • land abandonment

Introduction

Meadows are characteristic features of the forested mountain landscape in the northern temperate zone. Their ecological significance is substantial due to high biodiversity and the ecosystem services they provide: (1) provisioning services – source of food and forage, genetic resources, medicinal value, bioenergy production, (2) regulating services – soil erosion protection, water flow regulation, carbon sequestration, pest control, pollination, (3) habitat services – maintenance habitat for migratory species, gene pool protection, (4) cultural services – recreational and aesthetic value, inspiration for art and design (Hönigová et al. 2012). At the present time, forest encroachment is observed across many mountain meadows in the northern temperate zone. In effect, their area is gradually decreasing (Parolo et al. 2011).

Woody plant encroachment affects meadows along the entire altitudinal gradient: (1) meadows above treeline – in subalpine and alpine belts (Franklin et al. 1971; Woodward et al. 1995; Didier 2001; Sitko & Troll 2008; Leonelli et al. 2011; Martazinova et al. 2011; Zald et al. 2012; Durak et al. 2013), (2) montane meadows (Ciurzycki 2004a; Dovčiak et al. 2008; Halpern et al. 2010; Kucharzyk & Augustyn 2010; Rice et al. 2012; Tokarczyk 2013, 2015), (3) meadows below the forest-field boundary (Szwagrzyk et al. 2004; Frączek & Zborowska 2010; Zajíčková et al. 2011; Gallay et al. 2013) (Tab. 1). For example, since the early 1950s the total area of non-forest vegetation patches in the Central Western Cascade Range of Oregon (USA) has decreased by 54% (Takaoka & Swanson 2008) and by 66% in the Oregon Coast Range (USA) (Zald 2009). In a similar time period, there has been a decline in the area of montane meadows by 45% in the Tatra Mts. (Bukowski 2009) and by 32% in the Gorce Mts. (Tokarczyk 2012a) – both in the Polish Carpathians. All of the above mentioned studies were conducted at a similar elevation not exceeding 1,650 m.

Table 1. Changes in the extent of forest and non-forest areas in Europe with a special focus on the Carpathians

Author	Study area	Country	Time period	Object of interest	Changes in area/ elevation
Ciurzycki (2004a)	Tatra Mts. (Western Carpathians)	Poland	1977-1994	upper montane meadows	- 40.9%
Tokarczyk (2012a)	Gorce Mts. (Western Carpathians)	Poland	1954-2003	upper montane meadows	- 32.4%
Wężyk & Pyrkosz (1999)	Gorce Mts. (Western Carpathians)	Poland	1954-1997	montane meadows	- 21.6%
Ostapowicz & Szablowska-Midor (2005)	Babia Góra Mt. (Western Carpathians)	Poland	1965-1998 1977-1998	montane meadows	- 8.4% (S slope) - 32.9% (N slope)
Bukowski (2009)	Tatra Mts. (Western Carpathians)	Poland	1955-2004	montane meadows	- 45.4%
Kucharzyk & Augustyn (2010)	Bieszczady Mts. (Eastern Carpathians)	Poland	1886-2004	montane meadows	- 73.4%
Paterek & Olędzki (2005)	Tatra Mts. (Western Carpathians)	Poland	1977-1999	montane meadows	- 32.6%
				timberline	+ 45 m (Western Tatras) + 26 m (High Tatras)
Kucharzyk & Augustyn (2008)	Bieszczady Mts. (Eastern Carpathians)	Poland	1852-1994	polonynas timberline	- 9.6% + 10 m
Sitko & Troll (2008)	Petros Mt., Chornohora (Eastern Carpathians)	Ukraine	1933-2001	polonynas timberline	- 26% + 80 m
Martazinova et al. (2011)	Ukrainian Eastern Carpathians	Ukraine	1930s-2000	polonynas timberline	- 24% + 13 m
Didier (2001)	Maurienne Valley (Northern French Alps)	France	1870-1990 1939-1990	woodlands woodlands at timberline	+ 40% (regional scale) + 25% (site scale)
				subalpine meadows	- 51% (site scale)

Author	Study area	Country	Time period	Object of interest	Changes in area/ elevation
Durak et al. (2013)	Bieszczady Mts. (Eastern Carpathians)	Poland	1980-2009	shrublands at timberline	+ 24.9% (Mała Rawka Mt.) + 15.1% (Wielka Rawka Mt.)
Weisberg et al. (2014)	Carpathians (entire mountain range)	Poland Slovakia Ukraine Romania	1930-2000	timberline	+ 56.9 m
Kaczka et al. (2015a)	Babia Góra Mt. (Western Carpathians)	Poland Slovakia	1964-2009	timberline	+ 16 m
Kaczka et al. (2015b)	Tatra Mts. (Western Carpathians)	Poland Slovakia	1955-2009 1949-2009	timberline	+ 15 m (Rybi Potok Valley) + 10 m (Mengusovská Valley)
Mihai et al. (2007)	Iezer Mts. (Southern Carpathians)	Romania	1986-2002	timberline	+ 1.5 m yr^{-1}
Treml & Chuman (2015)	Hrubý Jeseník & Giant Mts. (Sudetes)	Czech Republic	1936-2005	timberline	+ 0.30 m yr^{-1} (Hrubý Jeseník) + 0.43 m yr^{-1} (Giant Mts.)
Gehrig-Fasel et al. (2007)	Swiss Alps	Switzer-land	1985-1997	treeline	+ 37.9 m (mean) + 28 m (median)
Leonelli et al. (2011)	Aosta Valley (Western Italian Alps)	Italy	1901-2000 2000-2008	treeline	+ 115 m + 10 m
Frączek & Zborowska (2010)	Świerzowa Ruska village (Lower Beskids, Western Carpathians)	Poland	1940-2007	open areas	- 76%
Bezák et al. (2007)	Bukovské vrchy Mts. (Eastern Carpathians)	Slovakia	1949-2003	extensive grasslands extensive grasslands with trees	- 3.1% - 41.2%
Kuemmerle et al. (2008)	Eastern Carpathians (border region)	Poland Slovakia Ukraine	1986-2000	fallow lands & afforested areas	+ 16.1%
Kozak (2005)	Polish Western Carpathians	Poland	1933-1995 1985-2000	forest agricultural areas	+ 4% - 6-7%
Cocca et al. (2012)	Belluno Province (Eastern Italian Alps)	Italy	1980-2000	forest agricultural areas	+ 21% - 40%
Kozak (2003)	Orawa region (Western Carpathians)	Poland	1823-2001	forest	+ 15%
Kozak et al. (2007)	Carpathians (northern part)	Poland Slovakia Hungary	1987-2000	forest	+ 0.38% yr^{-1}
Gellrich et al. (2007)	Swiss Alps & Jura Mts.	Switzer-land	1980-1990	forest	+ 1.9%
Tasser et al. (2007)	Waltner Mähder Passeier Valley (Eastern Central Alps)	Italy	1956-2000	forest	+ 48.3%

Gradual overgrowing of mountain meadows leads to important changes in landscape structure and its functioning. The effects of this process are perceived in different ways depending on the environmental, economic, and social context. Forest encroachment leads to degradation of unique plant communities, loss of biodiversity, and reduction in the populations of open-land species (Pavlů et al. 2005; Wesołowska 2009). The diversity of habitats and species can increase in the initial phase of secondary succession, but in long-term period the landscape becomes more homogeneous, thereby reducing biodiversity (Ruskule et al. 2012). The preservation of meadows is important especially for large herbivores as well as for numerous bird species, which use these open spaces for resting, feeding and nesting (Sirami et al. 2007).

According to Kostuch (2004), overgrowing of meadows produces an unfavorable effect on the sanitary state of forest. It is related to the disappearance of habitats for arthropod predators and parasitoids that control populations of tree pests. Finally, forest encroachment may reduce the recreational and aesthetic value of the mountain landscape – also in terms of multi-sensory perception (Schüpbach et al. 2004; Tokarczyk 2012b). However, it is also possible to perceive secondary succession as a process leading to the restoration of natural habitats for numerous forest species (Navarro & Pereira 2012). Furthermore, reforestation may contribute to a reduction in soil erosion and probability of landslides (García-Ruiz et al. 1996; Bezák et al. 2007).

Consequently, it is very important to better understand the mechanisms and dynamics of forest encroachment on mountain meadows, especially in the context of the protection of natural resources, sustainable environmental management, and monitoring of biodiversity. In this paper, an attempt is made to provide a comprehensive analysis of the scale and the main drivers of forest encroachment on temperate mountain meadows. The paper also serves as a review of current trends in research as well as methods used to identify ongoing changes in the landscape.

Types of temperate mountain meadows according to their origin and ecology

The term meadow is used to describe a plant community formed by herbaceous plants with a large share of grasses or other graminoids, characterized by structure and composition determined by mowing and occasionally by livestock grazing (Grynia 1995; Szweykowska & Szweykowski 2003). In a very broad sense the term meadow is used to describe all permanent grasslands, including hay meadows and pastures (Grzyb 1996; Szweykowska & Szweykowski 2003). This simplified approach is also used in this paper, thus the term meadow refers to montane meadows as well as high-mountain grasslands of subalpine and alpine belts.

In terms of origin, temperate meadows can be classified as: (1) natural meadows – created and maintained by specific environmental conditions or wild herbivores, (2) semi-natural meadows – associated with long-term, extensive human activity, (3) improved (intensive) meadows – newly established, simple cultivations based on sown and highly productive forage grasses and legumes (Nowiński 1970; Grynia 1995; Hejcman et al. 2013). Due to the artificial nature of improved meadows and their continuous use, they are omitted in the subsequent parts of this paper.

Natural meadows

Natural meadows are found locally at sites, where environmental conditions prevent the growth and development of woody plants. This category includes meadows situated above the climatic treeline. Natural meadows may also occur below this line: in the subalpine zone, in wetlands, on floodplains, as well as on ridges with poorly developed soil cover and on very shallow soils surrounding rocky outcrops. Other sites, where natural meadows are found include areas frequently affected by natural disturbances such as windthrows, wildfires, landslides, avalanches. However, these meadows often represent only temporary successional stages (Grynia 1995; Pärtel et al. 2005; Hejcman et al. 2013; Dengler et al. 2014).

Semi-natural meadows

Semi-natural meadows are found at sites whose natural vegetation is forest (Dengler et al. 2014). These types of meadows were created as a result of cutting trees, burning or due to human activity following windthrows and wildfires. While they need the presence of a stabilizing factor such as mowing or grazing, they are formed mostly by native species, which have spread across the landscape over the centuries (Dahlström et al. 2013; Zarzycki & Korzeniak 2013). Semi-natural meadows also include communities formed by the self-sodding of abandoned arable lands (Nowak & Kostuch 1974; Dengler et al. 2014).

Meadows whose origin is difficult to determine

It is sometimes difficult to determine the origin of certain mountain meadows, as some of them were created by one factor and then maintained and possibly expanded due to other factors (Billings & Mark 1957). Particular types of mountain meadows with an unexplained origin are grassy balds found in the Southern Appalachians (USA) (Lindsay & Bratton 1979) and in the Oregon Coast Range (USA) (Magee & Antos 1992; Zald 2009) as well as polonynas in the Eastern Carpathians (Poland–Slovakia–Ukraine border region) (Winnicki 1999; Winnicki & Zemanek 2003). These are areas of naturally occurring treeless vegetation, located on a well-drained site below the climatic treeline in a predominantly forested region (Mark 1958). The formation and maintenance of these meadows remains a subject of debate and multiple contradictory hypotheses have been advanced (Hołowkiewicz 1885; Rehman 1895; Billings & Mark 1957; Zarzycki 1963; Jasiewicz 1965; Lindsay & Bratton 1979; Weigl & Knowles 2014). The most common hypothesis of the origin of polonynas is that they arose due to natural environmental factors and may have later expanded thanks to human impact (Winnicki 1999; Winnicki & Zemanek 2003). This hypothesis, although not accepted by everyone, appears to find confirmation in paleobotanic research (Ralska-Jasiewiczowa et al. 2006) as well as in historical sources that suggest the presence of polonynas in this region prior to the development of pastures (Augustyn 1993).

Drivers of forest encroachment on mountain meadows

The observed encroachment of trees and shrubs on mountain meadows is associated with a current shift in the global and regional processes that once limited woody plant establishment (Lepofsky et al. 2003; Rice et al. 2012). According to numerous studies, this phenomenon may be driven by both (1) natural and (2) human-induced factors (Butler 1986; Miller & Halpern 1998; Dyer & Moffett 1999; Bolli et al. 2007; Gehrig-Fasel et al. 2007). The most often mentioned natural drivers are related to climate change, whereas human-induced drivers concern mainly changes in land use. The relationship between the aforesaid drivers and their real effects on the environment remains unclear, as some changes occur simultaneously (Dyer & Moffett 1999; Takaoka & Swanson 2008; Weisberg et al. 2013; Treml et al. 2016). In other cases, the role of different factors can be modified over time (Butler 1986). It is also obvious, that changes in complex mountain ecosystems may reflect the influence of multiple factors (Norman & Taylor 2005). Furthermore, the significance of these factors may vary depending on local environmental conditions (Miller & Halpern 1998).

Natural drivers

High mountain ecosystems are considered particularly vulnerable to long-term climate change (Svajda 2008; Améztegui et al. 2010). Climate warming may serve as a trigger for the gradual upward shift of the treeline as well as for woody plant encroachment on subalpine meadows. This effect is confirmed by research conducted in the American Pacific Northwest – in the Cascade Range (Franklin et al. 1971; Rochefort & Peterson 1996; Miller & Halpern 1998) and in the Pacific Coast Ranges (Woodward et al. 1995), as well as in the European Alps (Leonelli et al. 2011) and in the Carpathians (Kaczka et al. 2015b).

Changes in temperature and precipitation that result in a shorter duration of the snow cover and longer growing season facilitate the development of woody plants above the treeline and increase in their density within the treeline ecotone (Franklin et al. 1971; Miller & Halpern 1998; Lepofsky 2003; Norman & Taylor 2005). In addition to general trends, it is important to note seasonal climate fluctuations, which may be correlated with a short-term improvement of conditions for seedling growth and survival (windows of opportunity). In this context, woody plant encroachment may take on an episodic pattern (Franklin et al. 1971; Norman & Taylor 2005). Moreover, the effect of climate change is dependent on local environmental conditions. For example, an increase in average annual precipitation may either increase or decrease the rate of forest encroachment depending on the moisture regime of the site (Miller & Halpern 1998; Norman & Taylor 2005).

It appears that the overgrowing of mountain meadows found at high elevations may be a response to climate change. However, it is important to note that not every case of treeline advance is associated with climate change (Mihai et al. 2007; Sitko & Troll 2008; Treml & Migoń 2015; Treml et al. 2016). In mountain areas with a long history of livestock grazing, for example in Central Europe, it is vital to differentiate two processes: (1) rise of the potential treeline and densification of tree stands within the treeline ecotone caused by climate change, (2) return of woody plants to areas found above the anthropogenic treeline, where the forest was removed by humans. The latter process is generally related to changes in land use. For example, only 10% of new forest areas found at elevations above 1,650 m in the Swiss Alps were identified as true upward shifts of the treeline, whereas 90% represents areas affected by secondary succession. Moreover, only 4% of the upward shifts were related to climate change (Gehrig-Fasel et al. 2007). The situation is similar in the case of Babia Góra Mt. in the Polish Carpathians. The progressive character of the timberline, which is demonstrated by a 16 m increase in altitude in the period 1964-2009, is most probably an outcome of the synergy of several factors, including global warming and the cessation of grazing and logging in this region (Kaczka et al. 2015a). Similarly, Treml and Chuman (2015) determined that the timberline in the Sudetes has been advancing upwards at a rate of 0.30-0.43 m per year during the last 70 years in response to abandonment of agriculture and overall temperature rise. However, the effect of land-use changes was more important, especially in the lower part of the treeline ecotone (Treml et al. 2016).

It is worth mentioning here that some researchers have also suggested that forest encroachment may be triggered by another natural factor – a significant reduction in the number of wild herbivores such as deer or elk (Moore & Huffman 2004; Coop & Givnish 2007). However, this factor appears to play an important role rather at local scale.

Human-induced drivers

As previously mentioned, agriculture as well as logging have influenced for centuries position of the treeline in many temperate mountain ranges. Moreover, the vast majority of meadows found at lower elevations were created by human activity, which makes them particularly unstable ecosystems (Zarzycki & Kaźmierczakowa 2006) – highly dependent on traditional agricultural practices (Gartzia et al. 2014). Once stabilizing factors such as grazing and mowing are removed, natural succession tendencies start to predominate (Michalik 1986).

According to research conducted on the Czoło glade in the Gorce Mts. (Polish Carpathians), the period necessary to complete overgrowth by dwarf shrub communities

(*Vaccinietum myrtilli*) and then young spruce forest (*Plagiothecio-Piceetum tatricum*) is 30-35 years (Michalik 1990). Similarly, Kozak et al. (1999) found, using aerial photographs, that the spontaneous development of forest took 30-40 years in the Western Beskidy Mts. (Polish Carpathians). Other research carried out on the Príslopy glade in the Pol'ana Mts. (Slovak Carpathians) determined that the succession period starting from an abandoned meadow to pure spruce forest lasts 30-50 years (Hrivnák & Ujházy 2005).

In many developed countries, an overall decrease in the area of semi-natural meadows has been observed in the last several decades as a result of two contradictory trends: agricultural intensification and land abandonment (Halada et al. 2011). The cessation of mowing and grazing and progressive transformation of meadows into forest occurs most rapidly in less favourable mountain and peripheral areas (MacDonald et al. 2000; Sitzia et al. 2010). This process has been observed in the European Alps (Gellrich et al. 2007; Tasser et al. 2007; Cocca et al. 2012), Carpathians (Kozak 2003; Kuemmerle et al. 2008; Munteanu et al. 2014) (Fig. 1), Pyrenees (Améztegui et al. 2010; Gartzia et al. 2014), and Balkans (Vassilev et al. 2011). Both the causes and duration of land abandonment vary substantially from one region to another. Sometimes changes may occur abruptly. In other cases, the process may progress gradually in conjunction with political, socio-economic, technological, demographic, or cultural changes (Bucała & Starkel 2012).

For example, an abrupt change in land use occurred after World War II as a result of the resettlement of Boykos and Lemkos from the border areas in southeastern Poland (Lach 1975;

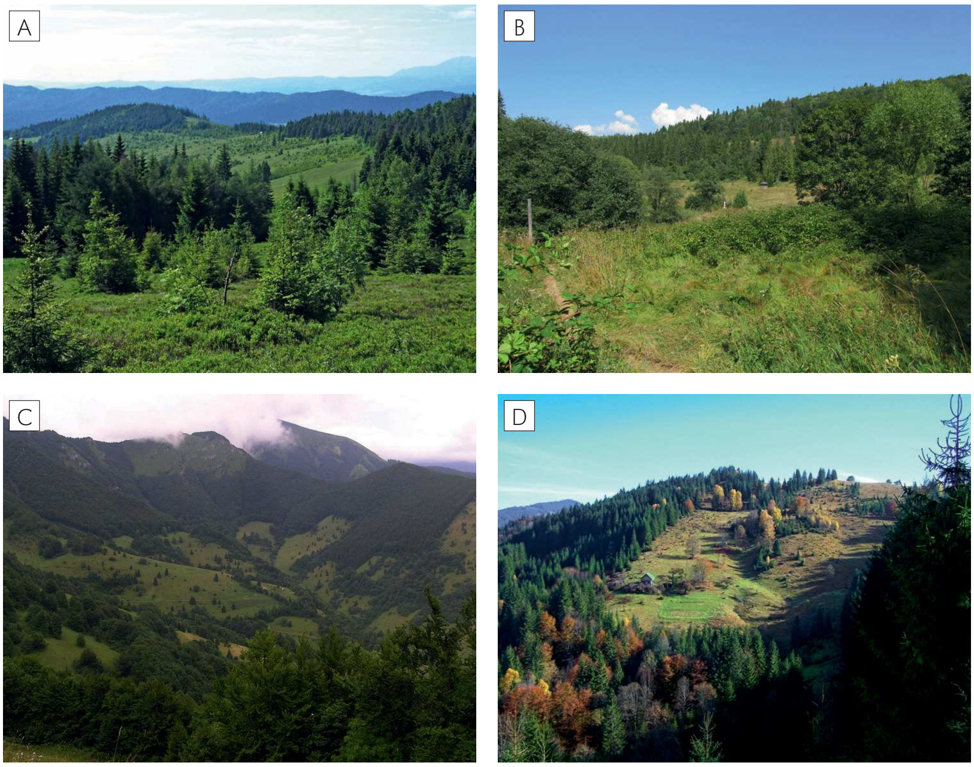

Figure 1. Examples of forest encroachment on mountain meadows in the Carpathians: A – Gorce Mts. (Poland), B – Bieszczady Mts. (Poland), C – Veľká Fatra Mts. (Slovakia), D – Chornohora Mts. (Ukraine) (Photo N. Tokarczyk)

Soja 2001; Wolski 2007) and the German minority from Poland and former Czechoslovakia (Prach et al. 1996; Pavlů et al. 2005; Latocha 2009; Zajíčková et al. 2011; Hejcman et al. 2013). Despite a number of later attempts, it was not possible to restore an extensive agriculture in these regions. In addition, closed military zones were also established in certain parts of Czechoslovakia (Levoča Mts., Doupov Mts.), where all civilian inhabitants were displaced and abandoned land became covered with forest (Dostalova 2009; Zajíčková et al. 2011). An abrupt cessation of agricultural practices may also be associated with the establishment of strict protected areas, as in the case of Tatra National Park in the Polish Carpathians (Ciurzycki 2003).

Gradual agricultural land abandonment began in many parts of Europe in the mid-19th century as a result of the Industrial Revolution (Gellrich & Zimmermann 2007; Leonelli et al. 2011). The rate of abandonment has further increased in the last several decades, although the exact rate does vary by region due to various historical and environmental considerations. The following factors are often cited as the main drivers for the decrease in agriculture in the mountains: mechanization and intensification of agriculture, urbanization of rural areas, depopulation – especially in remote areas, new sources of income, and land use limits associated with nature conservation (Miller & Halpern 1998; Bezák et al. 2007; Soliva 2007; Mladenov & Ilieva 2012; Munteanu et al. 2014; Nowak & Tokarczyk 2014). In some cases, spontaneous forest succession is supplemented by planned afforestation – especially of inaccessible, remote meadows.

Although intensive grazing limits competition from dense herbaceous vegetation, it may inhibit tree seedling establishment by browsing and trampling by livestock (Dunwiddie 1977; Van Uytvanck et al. 2008). According to Michalik (1990), the mesic meadow phase (*Gladiolo-Agrostietum deschampsietosum*) promotes spruce seedling establishment, but these are eliminated by grazing and mowing. A reduction in grazing pressure does facilitate woody plant encroachment, as competition from herbaceous vegetation is limited and seedling survival is enhanced at the same time. Moreover, trampling leads to moderate soil disturbances, which serve as a favourable microhabitats for seedling establishment (Dunwiddie 1977; Jurena & Archer 2003; Van Uytvanck et al. 2008). A complete cessation of grazing and mowing ultimately leads to forest encroachment on abandoned meadows. However, the rate of this process may change over time. In the period immediately following the cessation, when the sward is sparse and the soil is exposed in some places, woody plant encroachment may be facilitated. Dense sward formed over time as well as a thick layer of above-ground necromass affects seed access to soil moisture and nutrients, especially in the case of light-seeded species. Finally, a reduction in sward density in the subsequent phases of secondary succession (for example, through expansion of *Vaccinietum myrtilli*) accelerates woody plant encroachment (Lindsay & Bratton 1980; Michalik 1990; Frączek 1997, 2006; Van Uytvanck et al. 2008).

Forest encroachment on mountain meadows may also be related to decrease in the frequency of fires (Vale 1981; Norman & Taylor 2005; Takaoka & Swanson 2008). The regular occurrence of wildfires – particularly in more arid regions – has an significant impact on the maintenance of meadows (Magee & Antos 1992; Miller & Halpern 1998). Although the fires may be both natural (e.g. ignition from lightning strikes) and human-initiated, reduction in their number and extent results mainly from elimination of burning and suppression of wildfires, including development of the technology for fire detection and fighting. It appears that changes in fire frequency driven by climate fluctuations play in the above context a less important role. On the other hand, some researchers have argued that increased grazing pressure leads to the limiting of fire spread by the removal of fine fuels (Halpern et al. 2010), thus current land abandonment as well as reduction of landscape heterogeneity enhances fire risk (Rey Benayas et al. 2007).

Spatial scope and current research directions

Studies on forest encroachment on mountain meadows are conducted in many regions of the northern temperate zone. Most research is published in the United States, where forest succession has been examined for quite a long time (Lepš & Prach 1990; Szwagrzyk 2004). Case studies are based on multiple locations, including abandoned fields (old fields) on the East Coast (Bard 1952; De Steven 1991ab; Gill & Marks 1991), as well as mountain meadows on the West Coast (Franklin et al. 1971; Woodward et al. 1995; Miller & Halpern 1998; Takaoka & Swanson 2008). Studies conducted in the United States have produced a number of textbook examples of plant succession, although this 'American' process of succession cannot be fully translated into the European context due to different environmental conditions, species composition, and history of land use (Szwagrzyk 2004).

Either there exist fewer European studies on this subject or what is more likely – the research results are less accessible due to language barriers (i.e. non-English publications). Czech and Slovak researchers are known for producing a relatively large amount of research on this issue (Prach et al. 1996; Blažková & Březina 2003; Hrivnák & Ujházy 2005; Gömöry et al. 2006; Dovčiak et al. 2008; Dostalova 2009; Gallay et al. 2013). Forest encroachment on mountain meadows is also examined in the Polish Carpathians, especially in national parks (Bartoszek et al. 1990; Michalik 1990; Frączek 1997; Kuchnicka 1998; Bodziarczyk et al. 1999; Ciurzycki 2004abc, 2005; Szwagrzyk et al. 2004; Bukowski 2009; Frączek & Zborowska 2010; Kucharzyk & Augustyn 2010; Tokarczyk 2013). In research conducted in the European Alps, in addition to the classic approach to the encroachment of woody plants (Didier 2001), there is a new direction towards multi-factor modeling and forecasting of reforestation processes on a larger spatial scale (Gellrich et al. 2007; Tasser et al. 2007; Albert et al. 2008).

The overgrowing of mountain meadows has accelerated in the last few decades, which is why advanced research is needed using a variety of approaches. Research on forest encroachment on mountain meadows is pursued by researchers in fields such as botany and ecology as well as forestry, geoecology, GIS, and nature conservation. Despite fairly substantial progress in the understanding of applicable schemes and mechanisms, many questions remain to be answered. In addition, researchers are looking for supplementary data from other fields, while collaborative efforts are not numerous in the quest for a comprehensive and multidisciplinary approach (Lepofsky et al. 2003). The use of a single-perspective approach may lead to an incomplete picture of the problem. Research strictly focused on woody plant encroachment on various types of mountain meadows, which means the last stage of forest succession, may be divided into three groups.

The first group consists of studies on changes in meadow area, often in tandem with an attempt to determine rates of change and the calculation of basic landscape metrics, such as the degree of fragmentation or the number of forest patches (Wężyk & Pyrkosz 1999; Ostapowicz & Szablowska-Midor 2005; Zald 2009; Tokarczyk 2012a). The second group includes studies on the quantitative and qualitative characteristics of the said encroachment process (species composition and age structure of trees and shrubs as well as spatial patterns of encroachment) and sometimes also attempts to explain observed patterns (Lindsay & Bratton 1980; Bartoszek et al. 1990; Frączek 1997; Bodziarczyk et al. 1999; Didier 2001; Copenheaver et al. 2004; Szwagrzyk et al. 2004; Tokarczyk 2013). The third group consists of studies on factors affecting the rate and patterns of meadows overgrowth (Miller & Halpern 1998; Ciurzycki 2004ab; Andersen & Baker 2005; Dostalova 2009; Kucharzyk & Augustyn 2010; Bukowski 2013). Analyses of this type rely on various statistical methods ranging from simple statistics to model generation and change forecasting (Gellrich et al. 2007; Tasser et al. 2007;

Kucharzyk & Augustyn 2010; Bukowski 2013; Weisberg et al. 2013). Unfortunately, only a few such studies are available, however, it seems that their number has increased in recent years. The gap is still large in the case of montane meadows as well as research focused on a larger number of factors. Holeksa and Szwagrzyk (2006) do note that it is easier to simply identify changes without actually determining rates of change or explaining the mechanisms responsible for them.

Methods used to identify contemporary changes

Forest encroachment on mountain meadows is investigated on a variety of spatial scales ranging from experimental plots to entire geographic regions. The selection of materials and methods depends on the spatial scale of particular case study. Research can be divided into three groups based on spatial scale. The first group is composed of detailed research on a small spatial scale, conducted most frequently based on research plots – often located along predesignated transects. This makes it possible to collect phytosociological relevés (Lindsay & Bratton 1980; Michalik 1990; Bodziarczyk et al. 1992; Blažková & Březina 2003) as well as count trees and shrubs, and measure their physical size (Bartoszek et al. 1990; Magee & Antos 1992; Frączek 1997). Small-scale approaches often employ dendrochronological methods (Dunwiddie 1977; Woodward et al. 1995; Didier 2001; Copenheaver et al. 2004; Motta et al. 2006). By using tree rings researchers are able to identify periods when trees invaded individual meadows and determine the rates of forest encroachment. However, such methods are extremely time- and labour-consuming – both in the field and in the laboratory (Copenheaver et al. 2004). Moreover, they are invasive, and thus cannot be applied to every case. If a meadow is small enough, its full area can be examined in one study (Bartoszek et al. 1990). One limitation of this type of research is the risk of omission of certain relationships, which are dependent on environmental conditions and are observable only on a larger scale.

The second group consists of studies on midsize areas, for example, a catchment, a valley or a small mountain range. In this case, the basic method is comparison of maps from different years or chronological sequences of aerial photographs (Wężyk & Pyrkosz 1999; Takaoka & Swanson 2008; Bukowski 2009; Zald 2009, Tokarczyk 2012a). Unfortunately, the availability of aerial photographs is often limited and does not provide a continuous time series. However, it is worth noting that cadastral maps produced by Austrian cartographers in the mid-19th century contain among others information on land use, and therefore provide useful comparative information for areas occupied by the Austrian Empire in the 19th and early 20th centuries (Pancer-Koteja et al. 2006; Tasser et al. 2007; Kucharzyk & Augustyn 2010). Another important source of information is archival terrestrial photographs, including postcards. The repeat photography method remains uncommon, but it is a very valuable source of information, as it allows for a much longer analytical perspective than the use of aerial photographs as well as satellite imagery (Wrzesień & Zwijacz-Kozica 2006; Roush et al. 2007; Kaim & Kolecka 2010). The supplementary methods may also be classic field surveys and mapping with GPS.

The third group includes studies on a large scale – entire mountain ranges as well as geographic or historical regions. This type of research focuses on land use changes – especially changes in forest cover, however changes in meadow (open, non-forest) area are often indirect result of investigations. In this case, analyses are usually conducted using satellite imagery (Kozak 2005; Kozak et al. 2007; Kuemmerle et al. 2008) and are associated with a substantial degree of generalization and the risk of omitting small-scale changes. Moreover, there are new technologies that have some potential to quantify secondary succession across large geographic areas, including airborne laser scanning. For example, laser scanning in Szczawnica municipality in southern Poland has produced data, which indicate that about 40% of grasslands in the munticipality is experiencing secondary forest succession (Kolecka et al. 2015).

Finally, some researchers (Didier 2001; Tasser et al. 2007) have attempted to combine all three spatial scales and the methods associated with each of them. It is significant step in order to help better understand the temporal and spatial dynamics of forest encroachment on mountain meadows. One effective solution associated with this research approach is the analysis of a large sample of meadows that is representative of different environmental conditions (Miller & Halpern 1998; Szwagrzyk 2004; Takaoka & Swanson 2008).

Conclusions

Temperate mountain meadows are characterized by diverse origin, which means that both causes and rates of forest encroachment may vary in different regions. Decrease in mountain meadow area may be driven by natural factors related generally to climate change or may result from changes in land use. Mountain meadows are in many cases relatively small in size and surrounded by forest, which means that the colonization of these meadows by trees and shrubs is relatively easy. However, it is clear that natural meadows are less vulnerable to woody plants encroachment than semi-natural ones.

Studies on forest encroachment on mountain meadows are now conducted in many countries around the world and engage the efforts of many specialists in various scientific disciplines. However, there still does not exist a comprehensive, multidisciplinary approach. What is also lacking is comparative research analyzing many different mountain ranges, which would be highly useful due to the fact that some meadows remain free of woody plants for years, while others are colonized rapidly. It is difficult to explain such a diversity without large-scale experimental studies in a large number of meadows (Prach et al. 1996). In some cases, studies focused on the same geographic region may produce contradictory results when different methods are used or a different scale is applied. It is also possible, and this was noted by Szwagrzyk (1995), that an observed process cannot be included in a single model – only partial models explaining individual cases or small groups of cases are attainable.

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