

POST-SOCIALIST FOREST DISTURBANCE IN THE CARPATHIAN BORDER REGION OF POLAND, SLOVAKIA, AND UKRAINE

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Abstract. Forests provide important ecosystem services, and protected areas around the world are intended to reduce human disturbance on forests. The question is how forest cover is changing in different parts of the world, why some areas are more frequently disturbed, and if protected areas are effective in limiting anthropogenic forest disturbance. The Carpathians are Eastern Europe's largest contiguous forest ecosystem and are a hotspot of biodiversity. Eastern Europe has undergone dramatic changes in political and socioeconomic structures since 1990, when socialistic state economies transitioned toward market economies. However, the effects of the political and economic transition on Carpathian forests remain largely unknown. Our goals were to compare post-socialist forest disturbance and to assess the effectiveness of protected areas in the border triangle of Poland, Slovakia, and Ukraine, to better understand the role of broadscale political and socioeconomic factors. Forest disturbances were assessed using the forest disturbance index derived from Landsat MSS/TM/ETM+ images from 1978 to 2000. Our results showed increased harvesting in all three countries (up to 1.8 times) in 1988–1994, right after the system change. Forest disturbance rates differed markedly among countries (disturbance rates in Ukraine were 4.5 times higher than in Poland, and those in Slovakia were 4.3 times higher than in Poland), and in Ukraine, harvests tended to occur at higher elevations. Forest fragmentation increased in all three countries but experienced a stronger increase in Slovakia and Ukraine (~5% decrease in core forest) than in Poland. Protected areas were most effective in Poland and in Slovakia, where harvesting rates dropped markedly (by nearly an order of magnitude in Slovakia) after protected areas were designated. In Ukraine, harvesting rates inside and outside protected areas did not differ appreciably, and harvests were widespread immediately before the designation of protected areas. In summary, the socioeconomic changes in Eastern Europe that occurred since 1990 had strong effects on forest disturbance. Differences in disturbance rates among countries appear to be most closely related to broadscale socioeconomic conditions, forest management practices, forest policies, and the strength of institutions. We suggest that such factors may be equally important in other regions of the world.

Key words: Central and Eastern Europe; forest disturbance index; forest fragmentation; illegal logging; Landsat; land use and land cover change; post-socialist transition; protected areas, effectiveness; remote sensing.

INTRODUCTION

Anthropogenic land use is a major driver of change in terrestrial ecosystems and has modified more than half of the Earth's land surface (Vitousek et al. 1997, Foley et al. 2005). Forest ecosystems provide many structures and services that are essential for humanity, including the protection of biodiversity and carbon sequestration (Goodale et al. 2002, Randolph et al. 2005). Assessing changes in forest ecosystems and understanding their underlying causes is therefore of great concern. Global forest cover has been greatly reduced in the last centuries

(Goldewijk 2001), and continues to diminish, particularly in the tropics (Lepers et al. 2005). The extent (Skole and Tucker 1993, Achard et al. 2002) and underlying causes (Pfaff 1999, Geist and Lambin 2002) of tropical deforestation have received much attention. However, in other regions forests are increasing (Rudel et al. 2005), or forest cover trends are unknown, and a better understanding of forest cover change across the globe is needed.

Central and Eastern Europe still have large and relatively wild forests (Mikusinski and Angelstam 1998, Badea et al. 2004, Wesolowski 2005). The Carpathian mountain range presents Europe's largest continuous mountain forest ecosystem and is an important carbon pool, due to the high proportions of stands in higher age classes and the high productivity of Carpathian forests

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(Nijnik and Van Kooten 2006). Being a bridge between Europe's southwestern and southeastern forests, the Carpathians also serve as an important refuge and corridor for plants and animals (Perzanowski and Szwagrzyk 2001, Webster et al. 2001). The Carpathians harbor high levels of biodiversity with a large number of endemic species; over one-third of all European plant species (Perzanowski and Szwagrzyk 2001); and habitat for Europe's largest populations of brown bear (*Ursus arctos*), wolf (*Canis lupus*), lynx (*Lynx lynx*), wildcat (*Felis sylvestris*), and European bison (*Bison bonasus*) (Webster et al. 2001, Badea et al. 2004). Yet, relatively little is known about recent landscape changes in the Carpathians, and spatially explicit information on changes in habitat conditions is scarce.

Eastern Europe has experienced drastic changes in political, societal, and economic structures following the fall of the Iron Curtain in 1990. The transition from command economies to market-oriented economies had powerful impacts on land management and land use (GLP 2005), and resulted in forest cover change in many areas across Eastern Europe, for example in the Czech Republic (Bicik et al. 2001) or in Poland (Augustyn 2004). In areas where socialist forest management overexploited forests (Turnock 2002), forest cover has partially increased since 1990 (Peterson and Aunap 1998, Bicik et al. 2001). Conversely, privatization of forests may have increased harvesting rates (Eronen 1996, Turnock 2002) and illegal clear-cutting has occurred in some areas (Nijnik and Van Kooten 2000). We were particularly interested in assessing forest disturbance, which is the removal of forest cover by way of natural events (e.g., insect outbreaks, windfall) or anthropogenic activities (e.g., logging, infrastructure development). Little quantitative information on the rate and spatial pattern of disturbances in Eastern Europe's forest ecosystems is available for the post-socialist period. The question of how the political and economic transition affected forests remains, especially in the Carpathian Mountains where biodiversity is potentially threatened due to logging activities, which may lead to the fragmentation and degradation of forests.

Beyond the urgent need to assess forest disturbances in Eastern Europe, the region offers unique opportunities to better understand the role of socioeconomics for land dynamics (GLP 2005, Kuemmerle et al. 2006). Laws, policies, and institutions exert strong influence on land users and land management (Lambin et al. 2001, Dietz et al. 2003), and changes in broadscale socioeconomic and political determinants can trigger land change. However, the relative importance of broadscale factors on land cover dynamics is not well understood (GLP 2005). Land management policies and institutions in Eastern Europe changed dramatically after 1990. Assessing post-socialist land changes may thus reveal important insight into the effects of changing institutions on land cover (GLP 2005).

Cross-national studies in environmentally homogeneous regions are particularly interesting because they

allow relating differences in land dynamics to differences in socioeconomics and policies (Kuemmerle et al. 2006). The Carpathian Mountains are well suited for trans-border comparisons because the region is environmentally relatively homogeneous (UNESCO 2003), yet heavily dissected by country borders. The region was part of the Austro-Hungarian Empire for a period of ~150 years prior to 1918 (Turnock 2002), during which land management policies and land use were fairly homogeneous. However, in post-World War II socialist times, the Soviet Union and other Eastern European countries were distinctly different in politics and socioeconomics (Lerman 2001). After 1990, countries chose different approaches and rates in their transition to market-oriented economies (Lerman 2001). Comparison of post-socialist change in forest ecosystems (e.g., measured through disturbance rates) for border regions in the Carpathians thus offers unique opportunities to relate socioeconomic and political differences among countries to differences in land cover change.

Protected areas are important for conserving biodiversity (Myers et al. 2000), and several protected areas were established in the Carpathians to protect the region's unique forest ecosystems (e.g., UNESCO 2003). Protected areas face threats from human activities both within their boundaries and in their surrounding areas (Chape et al. 2005). Although protected areas stop habitat loss in most cases (Bruner et al. 2001), land use and land cover change in their neighborhood often reduces adjacent habitat (DeFries et al. 2005, Naughton-Treves et al. 2005), which is problematic for area sensitive species (Woodroffe and Ginsberg 1998). It is therefore crucial to quantify the effectiveness of protected areas and their management (Chape et al. 2005). This is commonly measured by comparing forest disturbance rates within protected areas and their neighborhoods (Bruner et al. 2001, Naughton-Treves et al. 2005). Transboundary protected areas are particularly interesting because forest disturbance rates inside and outside protected areas can be compared among countries. Differences between neighboring countries are likely due to differences in protected area management, institutions, and socioeconomic factors such as population density, rural income, or attitude toward protected areas. Cross-border comparison thus allows for a better understanding of the relative importance of broadscale determinants for the effectiveness of protected areas.

Comparing rates and spatial pattern of forest disturbances among countries in the Carpathians is not an easy task because conventional data sets such as forest inventory maps and statistical data are either missing or differ in scale and accuracy (Nijnik and Van Kooten 2000, Filer and Hanousek 2002). Moreover, illegal forest harvesting may be common (Nijnik and Van Kooten 2000), but is not included in official forestry statistics, thus limiting the use of such statistics. An alternative is to map forest disturbances using satellite

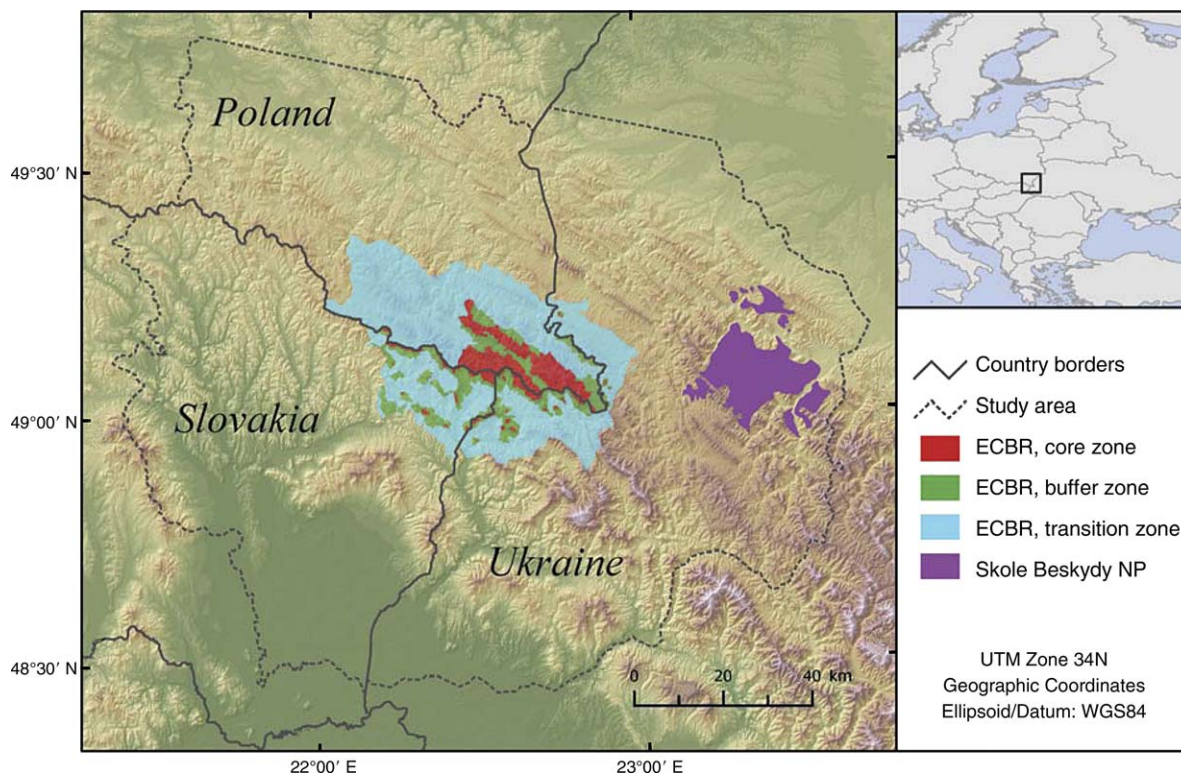


FIG. 1. Location of the study region in the Carpathian Mountain range, Eastern Europe. The study region harbors two protected areas, the trilateral East Carpathians Biosphere Reserve (ECBR) and the Skole Beskydy National Park (NP) in Ukraine (elevations range from ~100 to 1700 m; data sources: SRTM digital elevation model, ESRI Data and Maps Kit).

images (Coppin and Bauer 1996, Radeloff et al. 2000, Broadbent et al. 2006) because it provides current and retrospective land cover information, independent from country borders and in an efficient manner for large areas. The forest disturbance index (Healey et al. 2005) has recently been developed, but was so far only tested in the northwestern United States and in northern Russia. Landsat satellite data is particularly well suited for forest disturbance detection because of its relatively high resolution (80 m for Landsat Multispectral Scanner [MSS], and 30 m for Landsat Thematic Mapper [TM] and Enhanced Thematic Mapper Plus [ETM+]), and continuous data record since 1972, making it the most important data source for land cover change analyses (Cohen and Goward 2004).

Our study area was the border triangle of Poland, Slovakia, and Ukraine (Fig. 1). These three countries exhibited strong differences in socioeconomic and political determinants both before and after 1990, and this has affected forest ecosystems in our study area and resulted in differences in forest cover and forest composition among the countries. For example, the Ukrainian region of the study area has abundant coniferous forest whereas mixed and broad-leaved forests dominate in the Polish and Slovakian region of the study area (Kuemmerle et al. 2006). The question remains however, how much of such differences are due

to recent changes in the post-socialist period vs. pre-1990 socialist forest management. In other words, have the three countries converged since 1990 in terms of their forest cover and patterns due to the fundamental shift from a planning economy to a market-oriented system, or have they diverged?

The overarching goal of our study was to monitor post-socialist forest disturbance for the border triangle of Poland, Slovakia, and Ukraine in the Carpathians, because of the region's value for nature conservation and its high biodiversity, and because cross-border comparison of forest disturbance may also provide unique insights about the role of broadscale socioeconomic factors, policies, and institutions on land change.

Our specific objectives were thus to: (1) quantify post-socialist forest disturbance and make a cross-border comparison for parts of the countries Poland, Slovakia, and Ukraine in the Carpathians; (2) assess the effectiveness of protected areas in each country by comparing forest disturbance inside and outside protected areas; and (3) test the newly developed forest disturbance index in temperate mixed forests in order to measure forest disturbance between 1988 and 2000.

STUDY REGION

The study area covers 17700 km². Study region boundaries were based on administrative borders, the

extent of one Landsat TM scene, and landscape features such as rivers. Altitudes vary from 100 to >1300 m above sea level. The bedrock is largely dominated by sandstone and shale (Denisiuk and Stoyko 2000, Augustyn 2004), but some andesite-basalts occur in the southwest of the study area (Herenchuk 1968). With average annual precipitation of ~1200 mm and an annual mean temperature of 5.9°C (at 300 m), the climate is moderately cool and humid with marked continental influence (Augustyn 2004).

Our study area represents one ecoregion, but contains three altitudinal zones of potential natural vegetation (Perzanowski and Szwagrzyk 2001). The foothills (<600 m) are mostly covered by broad-leaved forests, consisting of European beech (*Fagus sylvatica*), pedunculate oak (*Quercus robur*), sessile oak (*Quercus petraea*), lime (*Tilia cordata*), and hornbeam (*Carpinus betulus*). The montane zone (600–1100 m) is dominated by European beech (*Fagus sylvatica*), mixed with silver fir (*Abies alba*), Norway spruce (*Picea abies*), sycamore (*Acer pseudoplatanus*), and white alder (*Alnus incana*) (Novotny and Fillo 1994, Grodzinska and Szarek-Lukaszewska 1997, Perzanowski and Szwagrzyk 2001). The timberline of dwarfed beech (1100–1200 m) directly borders alpine meadows on hilltops (Denisiuk and Stoyko 2000). The study area is environmentally relatively homogeneous (UNESCO 2003); however, local climate variations and topography result in a natural variability of forest types and forest composition (Denisiuk and Stoyko 2000). For instance mixed beech/fir forests are the natural vegetation on north-facing slopes, whereas pure beech forests would dominate south-facing slopes without anthropogenic influence. Forests in the study region are characterized by their high productivity, with annual increments in standing volume reaching up to 6 m³/ha (Nijnik and Van Kooten 2000, MASR 2003).

The study region harbors several protected areas (Fig. 1). The 29 000-ha Bieszczady National Park in Poland was founded in 1973 and enlarged several times until 1999. In 1992, the Polish–Slovakian biosphere reserve was designated consisting of Bieszczady National Park, two newly founded Polish landscape parks (San Valley and Cisniansko-Wetlinski), and the 46 000-ha Poloniny National Park in Slovakia. The biosphere reserve was transformed into the trilateral East Carpathians Biosphere Reserve when the Ukrainian Nadsanski Landscape Park (founded in 1997) and the Uzhanski National Park were joined in 1999 (Denisiuk and Stoyko 2000). The 39 000-ha Uzhanski National Park was also designated in 1999. Altogether, the East Carpathian Biosphere Reserve covers an area of ~213 000 ha (53% in Poland, 19% in Slovakia, and 28% in Ukraine). The biosphere reserve (Fig. 1) consists of a strictly protected core zone, a buffer zone (where conservation is emphasized, but sustainable land use and tourism are allowed), and a transition zone (where sustainable land use and development is promoted) (Denisiuk and Stoyko 2000,

UNESCO 2003). Another protected area, the 40 000-ha Skole Beskydy National Park, was established in 1999 in the Ukrainian region of the study area.

Although some of Europe's last remaining primeval forests are found in the study area, forest management has a long tradition in the region (Novotny and Fillo 1994, Augustyn 2004), and intensive land use has substantially affected most forests, creating a complex pattern of forests, arable land, and pastures (Grodzinska and Szarek-Lukaszewska 1997, Denisiuk and Stoyko 2000, Kuemmerle et al. 2006). Forest cover decreased markedly in the 18th and the first half of the 19th century due to population growth and land use intensification (Augustyn 2004). Since the 19th century, forest cover has generally increased (Kozak 2003). However, after World War II socialist forest management overexploited forest resources and logging rates again became unsustainably high in many areas (Turnock 2002). Some areas in the Polish region of the study area were depopulated after 1947 following border changes between the Soviet Union and Poland (Turnock 2002), and large areas were converted to forests (Augustyn 2004).

Forestry is an important factor for the local economy of the area (Antoni et al. 2000, Turnock 2002), and the majority of the forests in all three countries are used commercially. Most of the harvested timber is used to meet the demand of wood products in the respective countries and is not exported (Eronen 1996, MASR 2003). In Poland and Ukraine, harvested timber is mainly processed to sawnwood, particle board, used for paper and cardboard production, and to manufacture furniture (Andousypine 1994, Buksha et al. 2003, FAO 2005). In Slovakia, most timber is used for producing pulp for the paper and cardboard industry and for sawnwood (MASR 2003). Forest management has changed the forest composition in many areas and led to widespread replacement of natural forest ecosystems with Norway spruce and Scots pine monocultures (*Pinus sylvestris*) (Perzanowski and Szwagrzyk 2001, Augustyn 2004, Kruhlov 2005). The age compositions of forests in Poland and Slovakia are relatively close to an even distribution and most trees are found in mature age classes (Röhring 1999, MASR 2003). However, in Ukraine the age distribution is severely skewed toward young age classes, and <30% of all forests are mature (Strochinskii et al. 2001). The rotation age in commercially used forests varies depending on the species composition, but is on average around 80–120 years in Ukraine and 100–120 years in Poland and Slovakia (MASR 2003). Forest disturbance in the study region is largely anthropogenic, consisting mainly of logging and infrastructure development (Schelhaas et al. 2003). Natural disturbance events (e.g., insect defoliation, avalanches, and windthrow) are largely confined to plantations (Nilsson and Shvidenko 1999).

The transition from command to market oriented economies has affected the forestry sector and led to

changes in forest ownership, management policies, and institutions. In socialist times, nearly all forests in the study area were state owned (Turnock 2002), but forest management differed among countries. For example, clear-cuts were common in Ukraine and Slovakia, whereas selective logging dominated in the Polish region of the study area. After 1990, each country adopted a different transition strategy (Kissling-Naf and Bisang 2001), changing forest management and ownership patterns. Forests remained largely state owned in Ukraine and Poland, whereas Slovakia restituted forest to former owners (MASR 2003, FAO 2005). New forest management policies committed to multifunctional forestry were adopted in many Eastern European countries to comply with international agreements such as the Rio Protocol and the Helsinki Initiative (Kissling-Naf and Bisang 2001). In addition, Poland and Slovakia strived to meet European Union (EU) environmental standards in preparation for their accession to the EU (Eronen 1996). The demand for forestry products increased in Poland after 1992 and remained relatively stable in Slovakia, but has decreased considerably in Ukraine throughout the 1990s (Eronen 1996, MASR 2003).

Little quantitative information is available on how changes in forest ownership and forest legislation affected forest cover in the Carpathians. Official statistics are spatially coarse and overlook illegal forest activities. Remote sensing is the most feasible way to derive spatially explicit change information for large areas and across country borders. A few studies used remote sensing images to assess forest cover change in the Carpathians, but they were either restricted to small areas or relied on coarse resolution data (Kozak et al. 1999, Otahel and Feranec 2001, Kruhlov 2005). Coordination of Information on the Environment of the European Union (CORINE) 1:100 000 land cover data and Landsat MSS images showed an intensification of agriculture in Slovakian mountain valleys and a 9% loss in forest cover for the period 1976–1990 (Feranec et al. 2003). Historical maps and contemporary satellite images show increasing forest cover during the 20th century for several areas in the Carpathians (Angelstam et al. 2003, Kozak 2003, Augustyn 2004). Comparison of global land cover maps (at 1-km spatial resolution) for sub-catchments of the Tisza River in Ukraine showed a mean forest loss of 5% from 1992 to 2001 (Dezso et al. 2005). To our knowledge, no study has quantified Carpathian forest cover change for the post-socialist period at sufficient spatial detail and across borders.

DATA AND METHODS

Data sets used

We acquired five Landsat TM and ETM+ images (path/row 186/26, 10 June 2000, 4 July 1994, 2 June 1994, 27 July 1988, and 2 October 1986), and four Landsat MSS images (path/row 200/26, 30 July 1977; 200/25, 16 May 1979; 201/25, 2 September 1979; and

201/25, 2 July 1979). Thermal bands were not retained. The Space Shuttle Radar Topography Mission (SRTM, Slater et al. 2006) digital elevation model (DEM) was resampled to 30 m using bilinear interpolation to match the Landsat TM data. The borders of the protected areas were provided by the Geography Department of the Ivan-Franko University (Lviv, Ukraine).

To validate the accuracy of our forest disturbance map, ground-truth points were gathered in the field, from ancillary data sets, and from the Landsat images. Field work was carried out in summer of 2004, spring of 2005, and spring of 2006, using non-differential Global Positioning System (GPS) receivers. To cover broad areas and to avoid deterioration of the GPS signal under closed canopies, some areas were photo-documented from view points (e.g., mountain ridges). The view points were georeferenced using GPS receivers, and the view angle and distance of the area depicted in the photo were noted. This allowed digitizing ground-truth points on screen using the Landsat images and topographic maps as geometric references (Kuemmerle et al. 2006). Sixteen Quickbird and three IKONOS images (acquired between 2002 and 2005), and forest inventory maps and stand statistics from 1995 to 1999 for parts of Poland (obtained from the Polish Forest Administration) were used to collect additional ground-truth points. Clear-cuts frequently occurred in remote areas, for example away from roads or at higher altitudes, where mapping in the field was not feasible. To include these areas in our accuracy assessment, we digitized ground-truth points for bigger clear-cuts directly on the Landsat images. We included ground-truth points only where land cover was locally homogenous (i.e., 3×3 Landsat TM pixels) to minimize positional uncertainty and collected ~450 ground-truth points each in three categories: unchanged forest, non-forested, and forest disturbances. In total, 1347 control points were gathered (587 based on ground visits, 430 from ancillary data sets, and 330 from the Landsat data).

Preprocessing of Landsat TM and ETM+ data

Change detection requires precise geometric correction of images because misregistration and relief displacement decrease change detection accuracy (Coppin et al. 2004). We first referenced the June 2000 Landsat image to the Universal Transverse Mercator (UTM) coordinate system (World Geodetic System 1984 datum and ellipsoid), using the SRTM digital elevation model as a base map. To better match the June 2000 Landsat image, the SRTM DEM was shaded using sun azimuth and elevation from the Landsat acquisition date and time. Ground control points were collected semi-automatically using correlation windows (Itten and Meyer 1993, Kuemmerle et al. 2006). Once the June 2000 image was georeferenced, we co-registered all other satellite images to that image. Remaining positional errors were low (root mean square errors 0.16–0.26 pixels).

Removing atmospheric influence and differences in illumination due to topography can improve change detection accuracy (Song et al. 2001). We applied calibration coefficients to estimate at-satellite radiance (Chander et al. 2004) and a modified 5S radiative transfer model that incorporates a terrain-dependent illumination correction (Radeloff et al. 1997) to calculate surface reflectance. To prevent overcorrection in areas of low illumination, the Minnaert constant (Itten and Meyer 1993) was set to 0.75 for the October image. Comparison of neighboring spectra from shaded and unshaded hillsides and visual assessments confirmed successful atmospheric and topographic correction.

Forest disturbance detection

Mapping forest disturbance digitally provides quantitative change information and is more repeatable than visual image interpretation (Coppin and Bauer 1996, Coppin et al. 2004). Tasseled cap indices (Crist and Cicone 1984) are commonly used for change analysis (Collins and Woodcock 1996, Franklin et al. 2001, Wulder et al. 2006). This transformation reduces the data dimension while emphasizing forest related features (Dymond et al. 2002, Healey et al. 2005) and leads to higher change detection accuracies (Collins and Woodcock 1996, Healey et al. 2005). Based on tasseled cap transformation, the disturbance index (Healey et al. 2005) provides a single index identifying areas where forest cover declined. The index assumes that forests are characterized by high greenness and wetness components, whereas disturbances will display low greenness and wetness, but high brightness. The index requires masking out all non-forest areas. After normalizing the individual tasseled cap components to a mean of zero and a standard deviation of one, the disturbance index is calculated as brightness minus the sum of greenness and wetness. Categorical change maps result from multi-temporal classifications of the disturbance index images (Healey et al. 2005).

We applied the forest disturbance index in our study area. The 1986–1988 imagery was used to separate forest from non-forest. The MSS data from 1977 to 1979 were only used to determine if forest openings in the 1986–1988 imagery were clear-cuts (and forested in 1977–1979) or permanent openings. Post-socialist forest disturbances were assessed by calculating disturbance index images for 1988, 1994, and 2000, and conducting a maximum likelihood classification for the combined data (Fig. 2). Our satellite analysis can not distinguish between anthropogenic and natural disturbance, and we thus labeled all changed areas generically as disturbance, but it is important to note that the vast majority of these disturbances are due to forest harvesting because large-scale natural disturbances are rare (Schelhaas et al. 2003).

Separating forested and non-forested areas for 1988.—Separation of forest and non-forest can be challenging

for some forest classes when using single-date imagery. For instance, young broad-leaved forests and meadows can be spectrally similar in summer images. Phenology information inherent in multitemporal imagery allows us to distinguish such classes (Dymond et al. 2002, Zhang et al. 2003). We used unsupervised iterative self-organizing data analysis (ISODATA) to cluster the autumn image (2 October 1986) into 40 classes (Fig. 2). Clusters were labeled as forest, non-forest, or temporarily assigned to a mixed class if they were ambiguous. Mixed classes were further subdivided with ISODATA (using 10–20 classes) based on the summer image (27 July 1988), to assign all subclusters to the classes forest or non-forest. Water pixels were masked out using thresholds for the near and mid-infrared bands of the 1988 image. To exclude small areas that are functionally not forest (e.g., hedges, gardens, riparian buffers), we labeled all patches below a threshold of 30 pixels as non-forest. This threshold was derived based on high-resolution images and field visits.

Four Landsat MSS images from 1977 and 1979 together covered the entire study area and were used to check whether openings in 1988 represented forest disturbances or permanent clearings (Fig. 2). First, we identified all non-forest patches that were within larger forest patches in the TM-based forest/non-forest map as potentially disturbed areas. Ground-truth and visual assessment showed that all potential disturbances smaller than 21 TM-pixels were indeed disturbed areas, and no disturbances exceeded 1000 TM-pixels (90 ha). The remaining patches (>21 pixels and <1000 pixels) were subset from the MSS imagery while retaining the spatial resolution of the TM images. Second, this subset was subdivided into forest and non-forest pixels using ISODATA clustering for each MSS image. Because the overall number of pixels in each subset was low (between 0.03% and 0.71% of the study area), 10–20 classes were sufficient to accurately identify disturbed areas in 1988 and these disturbances were included in the forest class.

Detecting forest disturbances for the period 1988–2000.—The disturbance index (Healey et al. 2005) was calculated for each year (Fig. 2). Individual bands were stacked into a composite image, and a combination of unsupervised and supervised classifications was used to identify “unchanged forest,” “disturbance 2000–1995,” “disturbance 1994–1989,” and “disturbance before 1988.” We digitized 60 circular training areas (7 ha each) for unchanged forest based on the Landsat images, forest inventory maps, and expert knowledge. For each of the disturbance classes, between 22 and 27 of the larger disturbances were digitized on screen. All training data were independent from accuracy assessment data. Training polygons were clustered using ISODATA, and unambiguous clusters were used as training signatures for a maximum likelihood classification (guided clustering, Bauer et al. 1994). Additional training signatures were gathered interactively for areas where misclassifications occurred.

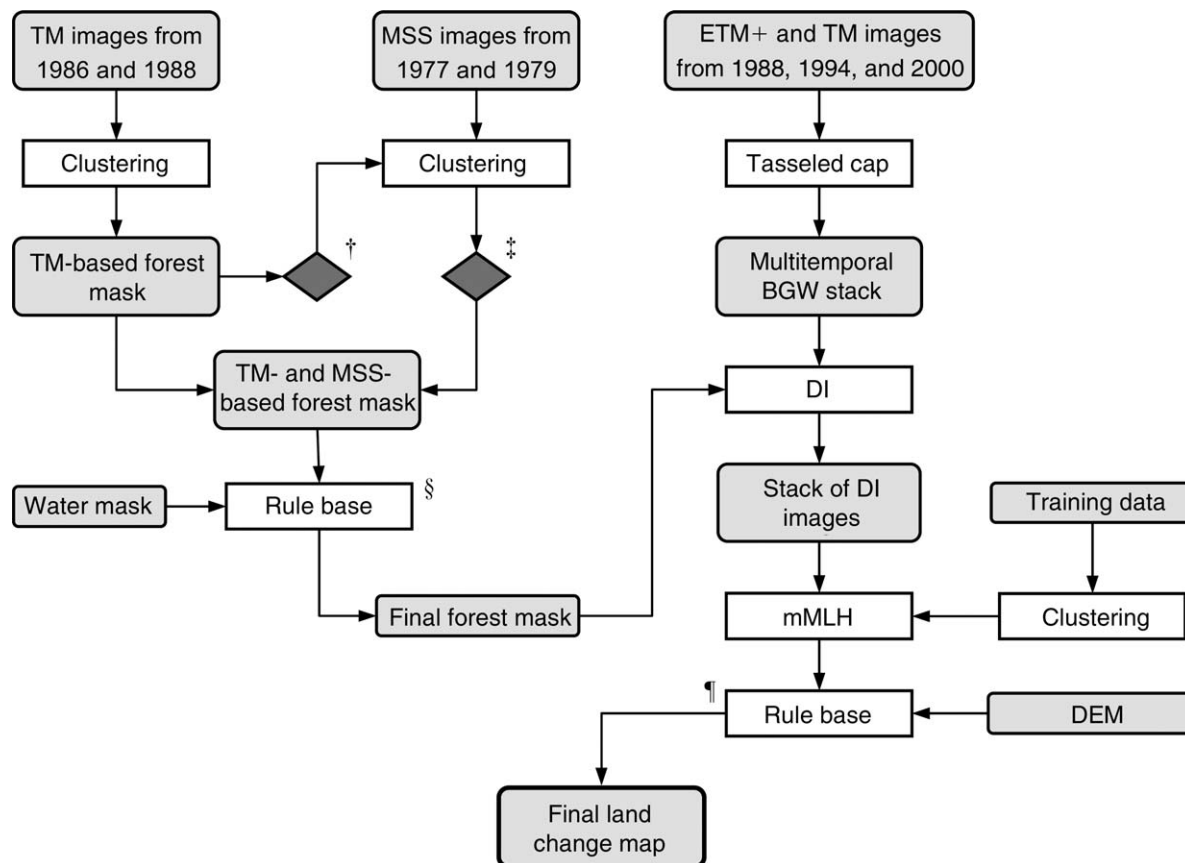


FIG. 2. Processing scheme for detecting forest disturbance in the study region (for details see *Data and methods: Preprocessing of Landsat TM and ETM+ data*). Abbreviations: DI, disturbance index; mMLH, multitemporal maximum likelihood; MSS, Landsat Multispectral Scanner; TM, Landsat Thematic Mapper; ETM+, Enhanced Thematic Mapper; BGW, tasseled cap brightness, greenness, and wetness; DEM, digital elevation model. Key to symbols: †, non-forest patches within bigger forest patches?; ‡, forest in 1977–1979; §, eliminate small patches and mask out water; ¶, eliminate small patches and misclassifications at higher elevations and on the forest fringe.

The TM images from 1994 and 1988 contained a few clouds (0.9% and 2.2% of the study area, respectively). For those areas, disturbance index images were calculated from additional images. The 1988 image was substituted with an image from 1986, whereas for 1994 two images were available. Because the area affected by clouds was very small for 1994 and 1988, thresholds proved to be sufficient to separate changed from unchanged areas. Some errors of commissions of disturbances occurred at elevation higher than 1050 m, due to phenological differences between the images, and these areas were labeled as unchanged. To remove noise due to misclassifications, patches smaller than seven pixels were eliminated (treating all forest disturbances as a single class to retain heterogeneity among disturbance classes) and assigned to the dominant surrounding land cover of either non-forest or unchanged forest. The threshold was determined based on visual assessment of very-high resolution images and ground truth. Some misclassifications occurred at the forest fringe (typically 1–2 pixels wide). Such patches were selected based on

their geometry and neighborhood characteristics and assigned to either forest or non-forest based on the disturbance image of 2000.

Disturbance data was summarized for the three periods covered by the Landsat TM/ETM+ data (before 1988, 1988–1994, and 1994–2000). We calculated annual disturbance rates by dividing the disturbed area for a given time period by six, thereby assuming disturbances detected in 1988 also had occurred in a six-year period. To compare forest disturbances inside and outside protected areas, disturbance rates were calculated separately for each of the protected areas and outside protected areas for each country.

Forest type stratification for changed areas.—To assess the type of forest affected by disturbances, we stratified 1994 and 2000 disturbed areas into broad-leaved forest, mixed forest, and coniferous forest based on the tasseled cap transformed 1988 Landsat image. To evaluate the accuracy of the forest type classification, we also included some areas of unchanged, mature forest where ground truth had been mapped (Kuemmerle et al. 2006),

and we used a stratified random sample of 250 such plots. We clustered the combined data set using ISODATA into 30 classes, which were labeled using expert knowledge and independent field data. Clouded areas in the 1988 image were classified using the same approach, but based on the 2 October 1986 image. Statistics were calculated based on the disturbed areas only. Disturbances in 1988 were not stratified into forest types due to the lack of ground-truth data for the MSS images.

Forest fragmentation

Forest fragmentation may introduce edge effects, lead to habitat loss, and result in a loss of forest biodiversity (Gascon and Lovejoy 1998, Debinski and Holt 2000, Riitters et al. 2002). Traditional landscape indices (O'Neill et al. 1988) and spatially explicit fragmentation measures (Riitters et al. 2002) can quantify forest fragmentation. We calculated the mean patch size and the area-weighted mean patch size of all disturbance patches for the three countries to examine forest disturbance sizes. The area-weighted mean patch size equals patch area (square meters) divided by the sum of patch areas (McGarigal 1994). To exclude micro-patches from the analysis, the forest disturbance map was majority filtered using a kernel size of 3×3 . To quantify changes in forest fragmentation, we used Riitters et al. (2002) indices. Riitters indices compare the proportion of forest (Pf) and forest connectivity (Pff) in a window around each pixel. Pff is an approximation of the probability that a forest pixel is located next to another forest pixel (Riitters et al. 2002). Each pixel was categorized as either "core" ($Pf = 1$), "perforated" ($1 > Pf \geq 0.6$ and $Pf > Pff$), "edge" ($1 > Pf \geq 0.6$ and $Pf \leq Pff$), or "patch" ($Pf < 0.6$). We chose a neighborhood size of 9×9 pixels based on prior research (Kuemmerle et al. 2006).

RESULTS

The forest disturbance analysis revealed major changes in post-socialist times in all three countries (Fig. 3), but the nature and extent of changes differed markedly among countries and time periods. In Poland, disturbances were overall rare. Slovakia showed a heterogeneous pattern of disturbances stemming from both socialist times and the post-1990 transition period, particularly along the border to Poland. In Ukraine, disturbances were frequent and mainly clustered in the center and the northern slope of the Carpathians (Fig. 3).

Our classification of the forest disturbance index resulted in a precise forest disturbance map with an overall accuracy of 94.8% and an overall kappa (Foody 2002) of 0.93, and conditional kappa values above 0.95 for all three periods. Producer's accuracy was equally high, with the exception of 1988 where accuracy was 81%, mainly due to confusion with unchanged forest (Table 1). Forest was the dominating land cover type in the region covering 51% in 1988. At higher altitudes,

forest cover was much higher, increasing to almost 100% cover above 800 m. Below 800 m, forest cover was much lower in Ukraine compared to Poland and Slovakia, particularly at altitudes between 400 and 800 m.

In total, 510 km² of forest were disturbed (2.89% of the total forest area), and 353 km² (2.00% of the total forest area) of the disturbances occurred after 1988. Disturbance rates were generally moderate and similar trends occurred in all three countries. Disturbance rates increased in 1988–1994 compared to the last years of socialist management (by a factor of 1.3–1.8). Between 1994 and 2000, yearly disturbance rates declined markedly below pre-1990 values in all three countries (Fig. 4).

While the general disturbance trends of the three countries were comparable, we found distinct differences in the extent and the rate of disturbances. Annual disturbance rates were lowest in Poland (e.g., annual disturbance rates from 1994 to 2000 of only 0.05%). In Slovakia and Ukraine annual disturbance rates were higher by a factor of 2.3–4.5 (Fig. 4) and highest in Ukraine (up to 0.58%). In total, only 2.2% (55.5 km²) of the forested area was affected in Poland compared to 6.2% (144.2 km²) and 6.7% (310.6 km²) in Slovakia and Ukraine, respectively (Fig. 4).

Most disturbances in Poland and Slovakia occurred in the foothill zone (below 600 m), but the majority of disturbed forests in Ukraine occurred in the montane zone (between 600 m and 1100 m) (Fig. 5). The distributions of disturbed forests differed markedly from the distribution of total forest (unchanged and disturbed forests), and elevational distributions remained constant over time. In Poland disturbance was relatively more common between 300 and 500 m and less common above 600 m. In contrast, in Ukraine the disturbances were relatively more common at higher elevations. Only in Slovakia were the elevational distributions of forests and disturbances similar (Fig. 5).

Ukraine had by far the most extensive disturbance in all time periods with area-weighted mean patch sizes of 4.8–9.3 ha, which was 1.5–3 times bigger than in Poland or Slovakia (Fig. 3). Poland had the smallest disturbances, but area-weighted mean patch size increased from 1.7 to 4.0 ha in the 1990s. In Slovakia and Ukraine on the other hand, disturbances were larger in the 1988–1994 period (5.7 and 9.3 ha in area-weighted mean patch size, respectively) than in 1994–2000 (3.0 and 4.9 ha, respectively). Average disturbance size was always smaller than the area-weighted mean patch size due to many small disturbances.

The stratification of disturbances into forest types had an overall accuracy of 82.4% and user's accuracies of 88%, 67%, and 88% for broad-leaved, mixed, and coniferous forest, respectively. In Poland and Slovakia, the majority of disturbances occurred in broad-leaved forest (up to 74% and 95%, respectively). In Ukraine, the proportion of disturbed coniferous forests was much higher (up to 40% in 2000). Comparing the distributions

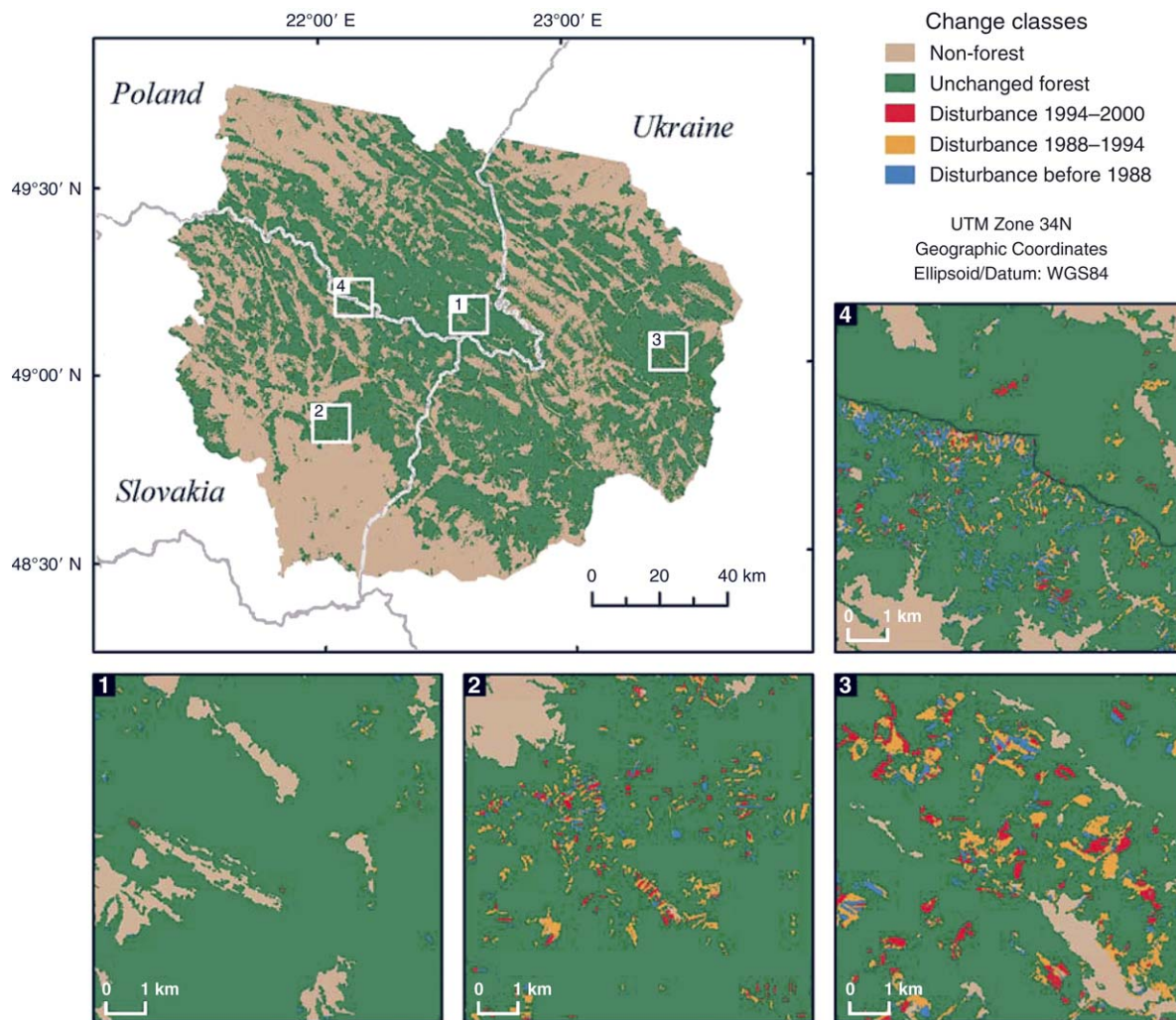


Fig. 3. Forest disturbance map of the study region. The insets provide examples of disturbance patterns of the countries (1) Poland, (2) Slovakia, (3) Ukraine, and (4) the Polish–Slovakian border region.

of disturbed forests over time, Poland and Slovakia showed little variation, whereas the Ukrainian share of coniferous forests increased from 28% to 40% (Fig. 6).

Higher disturbance rates in post-socialist times led to an increase in forest fragmentation in all three countries (Fig. 6). Core forest area decreased relatively little in

Poland (2.9%) compared to Slovakia (4.8%) and Ukraine (5.2%), where losses in core forest were connected to an increase in edge forest (3.0% in Slovakia and 3.6% in Ukraine). Generally, Poland had much higher shares of core forest and low levels of perforated forest (<5%), while Slovakia showed the lowest rates of

TABLE 1. Error matrix for the forest disturbance detection.

Classified data	Reference data					Σ	UAC
	NF	F	D2000	D1994	D1988		
Non-forest (NF)	440	10	5	3	7	465	94.6
Unchanged forest (F)	7	431	12	2	13	465	92.7
Disturbances in 1994–2000 (D2000)	0	1	194	3	0	198	98.0
Disturbances in 1988–1994 (D1994)	0	1	2	120	1	124	96.8
Disturbances before 1988 (D1988)	0	1	0	2	92	95	96.8
Σ	447	444	213	130	113	1347	
Producers accuracy (PAC)	98.4	97.1	91.1	92.3	81.4		
Conditional kappa	0.92	0.89	0.98	0.96	0.97		

Note: Values represent absolute numbers of ground-truth plots; UAC, user's accuracy (%); PAC, producer's accuracy (%).

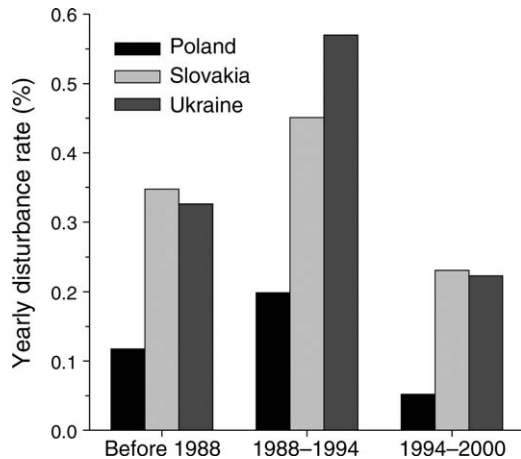


FIG. 4. Yearly disturbance rates for the Polish, Slovakian, and Ukrainian region of the study area.

Note: disturbance rates before 1988 were referenced to a six-year interval.

core forest and the highest shares of perforated and patch forest (Fig. 6).

Protected areas exhibited generally lower forest disturbance rates than non-protected areas, but this response varied strongly in time and among countries (Fig. 7). Poland generally had less disturbance than the other two countries in all zones, and the core zone was almost undisturbed in all time periods (maximum annual disturbance rate of 0.02%). Disturbances in the

buffer and transition zone were most frequent in 1988–1994, and it was surprising that annual disturbance rates in the buffer zone exceeded those outside protected areas (Fig. 7). In Slovakia, the core zone experienced much lower annual disturbance rates (up to nine times lower) than all other zones. Forest disturbance rates in the buffer and transition zones were higher than those outside protected areas before 1988 (annual rates >0.4%), but did not increase from 1988 to 1994 (unlike disturbance rates outside protected areas). From 1994 to 2000, rates dropped markedly, well below the annual rate of disturbances outside parks (Fig. 7).

In Ukraine, all zones of the protected areas experienced relatively high disturbance rates and annual rates inside protected areas were not substantially lower than those outside parks (Fig. 7). Unlike Poland and Slovakia, disturbances in the core zone in Ukraine increased, particularly in 1994–2000. In the transition zone and in the Skole Beskydy National Park, annual rates roughly doubled in 1988–1994 and exceeded disturbance rates outside protected areas (reaching annual disturbance rates of 0.86% and 0.65%, respectively), but rates decreased in 1994–2000 (Fig. 7).

DISCUSSION

Comparison of post-socialist forest disturbance rates among countries

Major changes in forest cover and forest fragmentation occurred in the border triangle of Poland, Slovakia, and Ukraine. Large-scale natural disturbances are rare

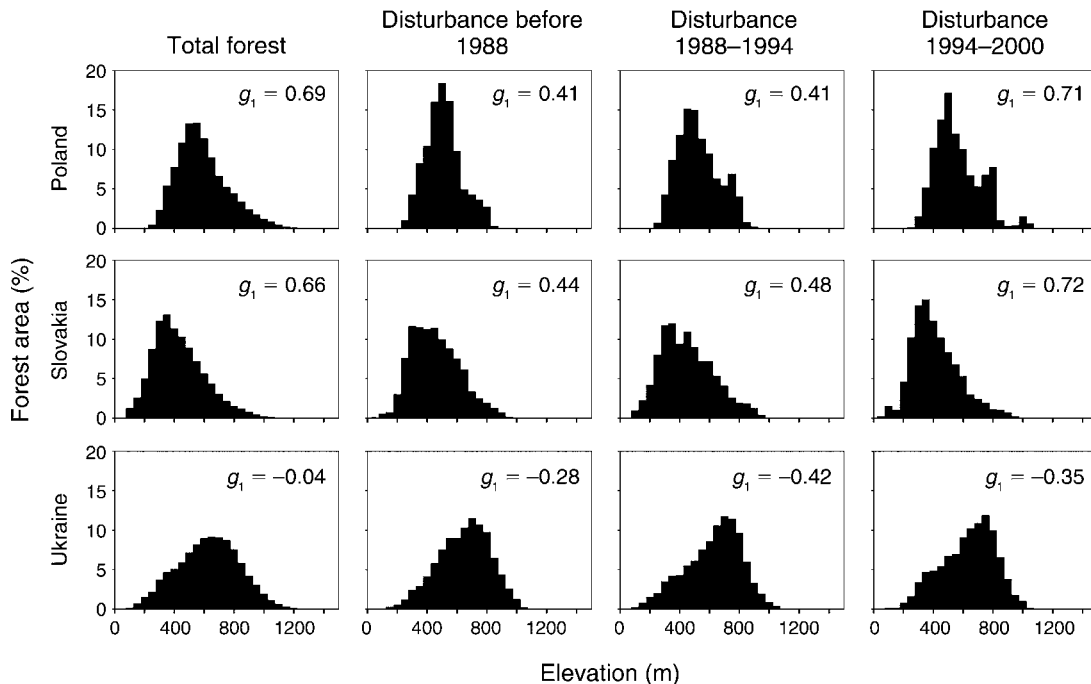


FIG. 5. Altitudinal distribution of total forest area (unchanged forest and disturbances) and disturbances for 1988, 1994, and 2000 for the three countries. (Distributions are normalized; g_1 = skewness.)

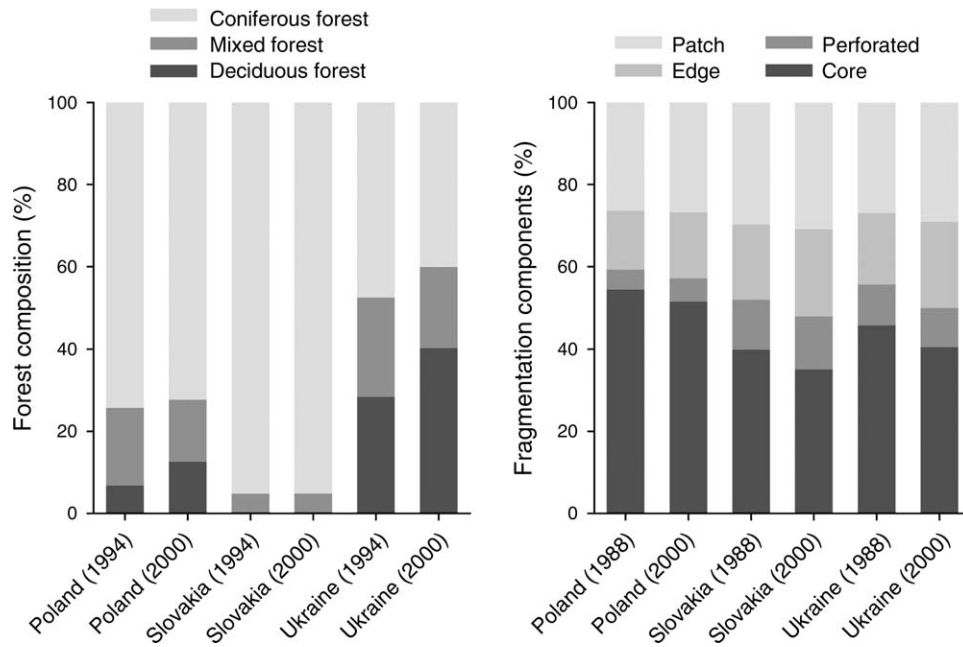


FIG. 6. Left: distribution of disturbed forests among the forest types broad-leaved forest, mixed forest, and coniferous forest for disturbances mapped in 2000 and in 1994. Right: forest fragmentation components for the years 1988 and 2000.

in the study area and most disturbances detected in our analysis can therefore be attributed to logging. Harvesting rates were relatively moderate overall and are not necessarily unsustainable considering the average rotation age (>100 years) in the region. However, the spatial pattern of disturbances revealed harvesting hotspots (e.g., the Skole region in Ukraine), where overexploitation likely occurs (Fig. 3). Trends in harvesting rates were similar in all three countries, and spiked markedly in the 1988–1994 period. We suggest that increasing rates are at least partially due to the fundamental changes in institutions, policies, and economic conditions during the transition from socialist to post-socialist regimes.

Poland had the lowest harvesting rates among the three countries (Fig. 4) and low levels of forest fragmentation (Fig. 6). These patterns are likely due to forest management practices and socioeconomic conditions. Timber harvesting is based on selective logging, which was already carried out before 1990 (Turnock 2002). Thus, although timber is being harvested, it leads to lower disturbance rates because the canopy is only partly removed. Some areas in Poland were depopulated after World War II, resulting in a very low population density, lower local demand for forestry products, and lower anthropogenic pressure on forest resources (Augustyn 2004). After the system change (i.e., in 1988–1994), harvesting rates increased only moderately (Fig. 4). This is likely due to the stable ownership situation, the policy framework, and the strength of institutions in Poland. Forests in the study region were almost entirely owned by the state in socialist times and

ownership did not change substantially after 1990. Forest institutions were reformed relatively quickly (Polish Forestry Act 1991/1997, Kissling-Naf and Bisang 2001), and forest management further improved toward sustainable forestry during the 1990s (Turnock 2002), which is reflected in an almost even age class distribution of Polish forests (Röhring 1999).

Slovakia differed markedly and showed higher harvesting rates (Fig. 4) and the highest forest fragmentation (Fig. 6), likely due to forest ownership, forest management policies, and harvesting practices. Forest ownership patterns changed after 1990, when 43% of forests were restituted to private owners (Eronen 1996, FAO 2005). The reform of forest management agencies and policies was slow (Kissling-Naf and Bisang 2001), partly due to the complex ownership situation (Eronen 1996). These factors, together with the economic depression in the early 1990s, likely led to increased forest harvesting for rapid profit realization (Eronen 1996, Webster et al. 2001, Turnock 2002). However, increased harvesting does not necessarily lead to unsustainable use of forest resources. Forest composition of much of Slovakia's forests is relatively natural (Oszlanyi 1997), and the age class distribution of Slovakia's forests is near normal with a high proportion of mature forests (MASR 2003). Moreover, disturbance rates were overall relatively moderate, particularly when considering the high annual increment of up to 6 m³/ha. Timber harvesting in Slovakia is largely based on clear-cutting, which led to higher levels of forest fragmentation and disturbance rates compared to Poland (Fig. 4).

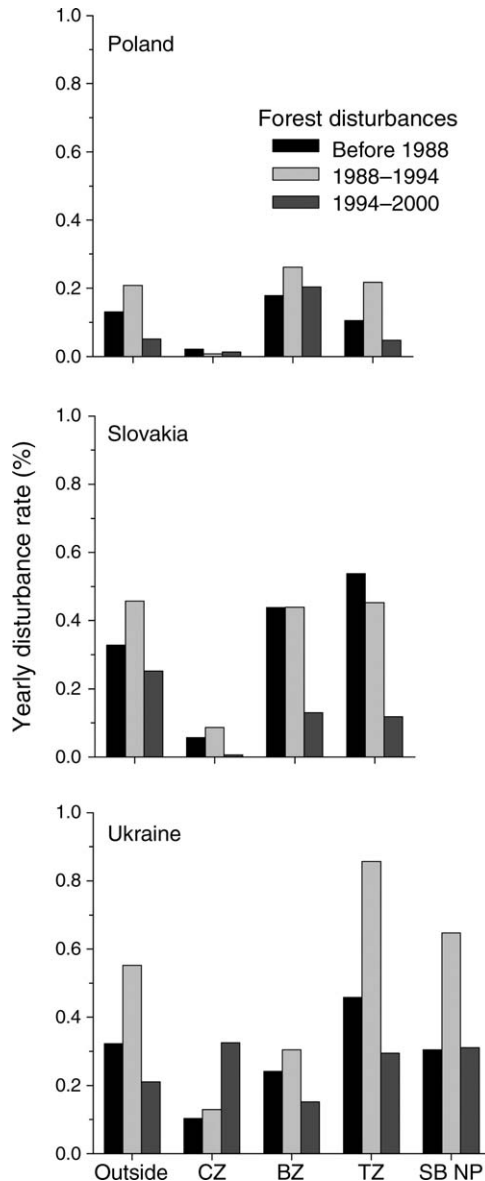


FIG. 7. Annual forest disturbance rates inside and outside protected areas per country and time period. Disturbance rates are given for the core zone (CZ), buffer zone (BZ), and transition zone (TZ) of the East Carpathian Biosphere Reserve, for the Skole Beskydy National Park (SB NP), and for areas outside of protected areas (outside).

In Ukraine, forest harvesting experienced the strongest increase in 1988–1994, but decreased below pre-1988 levels in 1994–2000 (Fig. 4). Forest ownership did not change after 1990 and all forests remained state owned (Turnock 2002). A new forest code toward more sustainable forestry was issued in 1994, but inadequate legislation and corruption resulted in a gap between policy and practice (Nijnik and Van Kooten 2000). After Ukraine became independent in 1991, administrative control decreased, but forest enterprises were still well equipped from Soviet times, funds were available, and

the wood processing industry was still active, altogether explaining higher harvesting rates. However, the general economic situation grew increasingly worse, and many forest enterprises did not modernize and became poorly equipped and funded (Turnock 2002). The demand for timber and the output of the wood processing industry fell dramatically (for example, –60% in sawnwood, –70% in particle board; Buksha et al. 2003) and both afforestation of farmland and reforestation after forest harvesting practically ceased (Nijnik and Van Kooten 2000, Buksha 2004). The age class distribution was already skewed toward younger ages due to heavy exploitation in socialist times, and mature forest became increasingly scarce (Nijnik and Van Kooten 2000, FAO 2005), which may explain decreases in harvesting between 1994 and 2000. The shortage of mature forest (<12% of total forests; Strochinskii et al. 2001) is also an explanation for harvesting of coniferous stands and at higher altitudes. Timber harvesting in Ukraine is generally based on clear-cuts using heavy machinery, thus explaining the bigger harvesting patches found there (Strochinskii et al. 2001).

Corruption and illegal forest harvesting in Ukraine increased during the transition phase, and this trend may continue in the future (Nijnik and Van Kooten 2000, Buksha 2004, Nijnik and Van Kooten 2006). Poverty is a driver of illegal logging (e.g., fuel wood harvesting; Turnock 2002), but there is also a substantial underground business in forestry (Nijnik and Van Kooten 2006) with largely unsustainable forest management practices. This is particularly apparent in the large volumes of so-called sanitary felling (i.e., clear-cuts of “unhealthy” stands), which reached 51% of all harvests in the Skole forestry district (Fig. 3, inset 3) between 1999 and 2005 (O. Chaskovskyy, *personal communication*). New forest policies place limits on clear-cuts of fir and beech forest on steep slopes, at higher altitudes, or in water protection zones, and envisage the increase of protected areas (Verkhovna Rada 2000a, b). It would be interesting to assess how these policy changes affected harvesting rates in Ukraine after 2000; however, this legislation does not effectively control sanitary felling practices.

Forest ownership pattern is important to understand forest cover change (Turner et al. 1996), but in our study area neither state forestry nor private forestry was clearly better in lowering harvest rates. Forests in both Poland and Ukraine are state owned, yet disturbance rates differed by a factor of 2.3–4.5. On the other hand, harvest rates in largely privately owned Slovakian forests were almost as high as in Ukraine. We found the highest harvest rates in the transition phase (1988–1994), and rates decreased where economies stabilized and after sustainable forest policies were launched. Thus, our results rather support the assumption that the strength of institutions is important and that good institutions result in stable or even increasing forest cover (Dietz et al. 2003, Tucker and Ostrom 2005).

Forest disturbances inside and outside protected areas

The marked differences in protected area effectiveness are likely related to socioeconomic conditions and strength of institutions. Protected area effectiveness was highest in Poland and Slovakia, whereas the establishment of protected areas in Ukraine lowered forest disturbance rates, yet, often not below harvest levels outside protected areas (Fig. 7).

Population density and poverty are drivers of anthropogenic forest disturbance (Lambin et al. 2001) and challenges for the effectiveness of protected areas (Naughton-Treves et al. 2005). In Poland, anthropogenic pressure on forest ecosystems is much lower compared to Slovakia and Ukraine, due to the depopulation of some areas in 1947. Harvest rates and forest fragmentation were very low (particularly in the core zone), and Poland had large continuous forest patches (Fig. 3). As a consequence, the highest densities of top carnivores and herbivores (e.g., wolf, brown bear, and European bison) are found in the Polish region of the study area (Perzanowski and Gula 2002). In Slovakia and Ukraine, population density is much higher and we found higher harvest rates inside protected areas (Fig. 7). However, the economic depression that occurred after 1990 lowered the effectiveness of protected areas in all three countries and forest harvesting increased from 1988 to 1994 within protected areas.

The designation of protected areas stops forest cover change in most cases (Bruner et al. 2001), even when institutions are weak (Naughton-Treves et al. 2005). This is supported by our results because harvest rates dropped markedly in all countries after protected areas had been established (i.e., in 1994–2000). Yet, the strength of institutions is another important factor for the effectiveness of protected areas. Poland and Slovakia have strong institutions and were on the eve of EU accession in the late 1990s. After parks were designated, harvest rates dropped well below rates outside protected areas, especially in Slovakia (Fig. 7). In Ukraine, where governance is not transparent and corruption is a problem (Nijnik and Van Kooten 2006), harvesting rates inside protected areas did not decrease below those outside protected areas, and were sometimes even higher. The weakness of institutions and park management is also apparent in the enforcement of park regulations (Bruner et al. 2001, Webster et al. 2001). Forest harvesting has caused increasing fragmentation inside and around protected areas in the Carpathians, similar to other regions in the world (Chape et al. 2005, DeFries et al. 2005, Naughton-Treves et al. 2005), which is especially problematic for top carnivores and herbivores (Woodroffe and Ginsberg 1998).

The age of protected areas can be an important determinant of park effectiveness because capacity building takes time. Protected areas in Slovakia, and particularly in Ukraine may be too young to draw final conclusions about the effectiveness of their park management. It is noteworthy though that forests in

Ukraine and Slovakia were heavily exploited immediately prior to the designation of protected areas, likely at the expense of biodiversity-rich older and near-natural forest in remote areas (Perzanowski and Szwagrzyk 2001). These fragmented large continuous forest patches and resulting edges effects may negatively affect forest biodiversity. Particularly in the Skole Beskydy National Park, where forest harvesting was concentrated (Fig. 3, inset 3), field visits in 2006 confirmed that logging is ongoing.

Comparison of forest disturbance rates and official statistics

Comparing our forest disturbance trends to official forestry statistics reveals agreement in some cases, and clear differences in others. In Poland, the amount of timber harvested was relatively stable according to statistical records in the last socialist years (Strykowski et al. 1993), and increased markedly throughout the 1990s (FAO 2005). Timber harvest statistics in Slovakia indicate a decline in the late 1980s from around $5.8 \times 10^6 \text{ m}^3$ to $<5 \times 10^6 \text{ m}^3$ between 1991 and 1993, but a considerable increase after 1993 to $>6 \times 10^6 \text{ m}^3$ in 2000 (Kolenka 1992, MASR 2003, FAO 2005). In Ukraine, harvesting trends are less clear. Some sources indicate decreasing harvesting in the 1990s (Nilsson and Shvidenko 1999, FAO 2005), yet, others show increased harvesting between 1986 and 1996 (Nijnik and Van Kooten 2000).

Several factors possibly explain differences between the statistics and the disturbance rates we derived from the remote sensing data. First, comparing harvested timber volumes (in cubic meters) and disturbed area is not easy because these parameters are not necessarily connected. For instance, increasing average stand age results in higher annual increments and standing volumes, thus allowing for increased timber harvests without automatically increasing the logged area. This may particularly be the case where the age class distribution of forest stands shows a high percentage of premature and mature stands such as for example in Slovakia (MASR 2003), and where sustainable forestry is in place (thus leading to a steady increase in standing volume). Conversely, if average stand age gradually decreases due to premature logging, a decline in timber volume harvested may still lead to an increase in disturbed area. Premature logging may be especially common where the age class distribution is skewed toward younger stands (e.g., in Ukraine; Strochinskii et al. 2001) and where new forest owners decided to realize returns quickly (Turnock 2002).

Second, selective logging is not detected with our methodology, yet, is the dominant harvesting practice in Poland. This inhibits the comparison of harvested timber volumes to our disturbance map because we defined disturbances as the complete removal of forest cover. Moreover, where forest management changes and selective logging becomes more common, for instance

due to policies that emphasize sustainable forestry (Kissling-Naf and Bisang 2001), the comparison of disturbance rates and timber volumes is difficult. Third, official statistics do not account for illegal logging, which is a particular problem in Ukraine (Nijnik and Van Kooten 2000, Buksha 2004), thereby underestimating actual disturbance rates. And last, the disturbance index may overlook some types of forest harvesting (e.g., very small clear-cuts). Although we cannot completely rule this out, our extensive accuracy assessment and field visits suggest a reliable forest disturbance map (see *Discussion: Accuracy of the forest disturbance detection* for details).

Accuracy of the forest disturbance detection

The disturbance index was so far only tested for three boreal study regions dominated by coniferous species (Healey et al. 2005). Our study was the first to apply the disturbance index to temperate forest ecosystems with mainly broad-leaved and mixed forest types. Overall, the disturbance index performed very well and the accuracy assessment confirmed an accurate change map.

The time interval between the images proved to be crucial for the successful mapping of forest disturbances. Due to the high productivity of Carpathian forests, vegetation regenerates quickly (particularly where reforestation is carried out) after a disturbance event. Thus, the disturbance index is most sensitive to relatively young disturbances, whereas the detection of older disturbances is difficult. The 1994 image was crucial in this respect because many post-socialist disturbances could not have been detected using 1988 and 2000 data alone.

Although our accuracy assessment confirmed the reliability of our change map, a few factors were identified that may have contributed uncertainty. First, reforestation of clear-cuts in Ukraine decreased dramatically after the system change (Buksha 2004). Later disturbances thus became easier to detect, because natural regeneration is slower. Disturbance rates from before 1988 may in such cases be underestimated. Second, the coarser spatial and spectral resolution of the MSS images compared to the TM/ETM+ data may have introduced uncertainty. However, it is important to note that the coarser-resolution data was only used to fill non-forest gaps in the initial TM-based forest/non-forest map. We included all non-forest patches smaller than 21 pixels (~1.9 ha) in our change analysis, to avoid an underestimation of pre-1988 disturbance rates in areas where clear-cuts were very small (e.g., in Slovakia). The change analysis was carried out using TM images only. The accuracy assessment, high-resolution images, and field visits did not suggest a systematic bias in our change map.

Third, the assumption that disturbances occur within forest patches may exclude disturbances at the forest fringe. Although we can not completely rule out that some disturbances were omitted, visual examination of

the Landsat images and additional high-resolution data showed that disturbances on the forest fringe were very rare, such that the effect seemed to be negligible. Fourth, phenological differences among the images may have affected disturbance detection. To accommodate for this, we did not apply uniform thresholds to determine changed areas, but used a composite classification, where phenological differences can be incorporated through appropriate training data for changed and unchanged areas. Nevertheless, phenology was a problem for some disturbances in 1988 that were spectrally similar to broad-leaved forest due to the late-summer image, and may have contributed to an underestimation of pre-1988 disturbance rates. Although differences in leaf onset in spring and defoliation in autumn may pose serious limitations when mapping forest disturbance of broad-leaved forests in mountain areas, this was not a problem in our case because we did not rely on leaf-off images. Last, the exclusion of forest disturbances smaller than seven pixels may have led to an omission of some very small clear-cuts, but we found that removing noise due to misclassifications had a much greater effect on the overall accuracy of the change map.

The disturbance index was unable to detect selective logging, where only a fraction of the canopy is removed; yet we were not interested in mapping such disturbances. Mapping selective logging sites may be important in other studies, and future research is needed to quantify the sensitivity of the disturbance index to detect selective logging.

To avoid an overly optimistic accuracy assessment, we used an equal sample for all classes (a random sample would place most control plots in stable forests, which are easiest to classify). Nevertheless, our accuracy assessment may be positively biased due to two factors. First, ground-truth plots were only established in locally homogeneous areas (3×3 pixels) to minimize misregistration error and to facilitate ground labeling (Foody 2002). This avoids class boundaries and mixed pixels, which can cause misclassifications (Foody 2002). Second, some disturbance plots were directly digitized from the Landsat data. Such an approach is common (e.g., Healey et al. 2005) because large disturbances can easily be identified. However, very small disturbances that are also harder to classify may be missed. We suggest that such errors were distributed evenly throughout the study area and among time periods, and did not affect the general differences among countries and disturbance trends that we observed.

CONCLUSIONS

Forest disturbances were frequent in the border region of Poland, Slovakia, and Ukraine in post-socialist times, and most disturbances represent forest harvesting because large-scale natural disturbance events are rare in the study region. Harvesting rates were generally relatively moderate; however rates increased in all three countries after the system change in 1990, leading to

higher levels of forest fragmentation. The increase in forest harvesting likely occurred due to ownership changes, worsening economic conditions, and the weakening of institutions. Forest disturbance rates differed markedly among countries, with much lower rates in Poland compared to Slovakia and Ukraine. We suggest that these differences can be explained by differences in forest management practices, forest policies, and the strength of institutions.

Protected areas generally exhibited less forest harvesting, but protection was far from complete, and the effectiveness of protected areas differed among countries. Protected area management was most effective in Poland, where population density is low and protected areas are relatively old, and in Slovakia, where harvesting rates dropped markedly below background levels after protected areas were designated. In Ukraine, harvesting rates inside protected areas were practically equal to those outside, and harvests were widespread immediately before the designation of protected areas.

Overall, the Polish, Slovakian, and Ukrainian regions of our study area have clearly diverged in terms of forest cover and forest fragmentation in post-socialist times. Poland, where forest cover was highest and forest fragmentation lowest, had the lowest disturbance rates. Conversely, Slovakia and Ukraine, with lower forest cover and higher forest fragmentation, had higher disturbance rates. While the stand age distributions of Poland and Slovakia do not necessarily suggest unsustainable use of forest resources, increased harvesting is of particular concern in Ukraine, where mature forests have become scarce.

The strong differences in harvesting rates that we found among the countries Poland, Slovakia, and Ukraine were determined by broadscale socioeconomic factors, past and present forest management practices, forest policies, and the strength of institutions. Cross-border comparisons can reveal important insights into the role of broadscale factors of human–environment interactions in forest ecosystems, and these factors may be equally important in other regions of the world.

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