

Humboldt-Universität zu Berlin – Geographisches Institut

**Studying land-use and land-cover change
with high resolution data – an assessment of the
Carpathian Ecoregion**

DISSERTATION

**Zur Erlangung des akademischen Grades
doctor rerum naturalium
(Dr. rer. nat.)**

Im Fach Geographie

**eingereicht an der
Mathematisch-Naturwissenschaftlichen Fakultät II
der Humboldt-Universität zu Berlin**

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Berlin, 07. Mai 2012 (Tag der Verteidigung)

Geography is a living, breathing subject, constantly adapting itself to change. It is dynamic and relevant. For me geography is a great adventure with a purpose.

So many of the world's current issues – at a global scale and locally - boil down to geography, and need the geographers of the future to help us understand them. Global warming as it affects countries and regions, food and energy security, the degradation of land and soils from over-use and misuse, the spread of disease, the causes and consequences of migration, and the impacts of economic change on places and communities. These are just some of the challenges facing the next generation, which geographers must help solve.

Michael Palin, The Guardian, August 2011

Acknowledgements

In writing, but most of all continuing and finally finishing this thesis, a great deal of people provided profound encouragement and belief along the way.

I would like to express my special gratitude to a few persons:

Patrick Hostert and Volker Radeloff for providing an exceptional relaxed but at the same time straight-forward research environment. Most importantly though, you owe my absolute esteem for slacken or tighten the reins at the right time, showing patients and foresight, and for providing me enough free space to come to the right decisions at the right time. Equally, I appreciate your scientific guidance through all steps of my work and doctoral training. Patrick, I'm deeply grateful for having the chance to be a member of the lab from almost the beginning and for receiving so much from it within all parts of life.

Tobias Kümmerle for your ongoing support, motivation and professional scientific guidance throughout the thesis and beyond.

Wilfried Endlicher for being a great teacher and having encouraged a diversified geographical thinking throughout my education at the institute.

Jacek Kozak for long-term Carpathian-science collaboration and evaluating this thesis.

My colleagues *Patrick Griffiths, Oliver Grübner, Andreas Rabe, and Anika Sieber* for all the work and support, which made important parts of the work at first possible.

My colleagues *Dirk Pflugmacher and Sebastian van der Linden* for important and profound last-minute proof-reading.

The overarching *Geomatics team* in general. The thesis would never have been possible as a stand-alone project. Moreover, the multi-coloured, multi-ethnic environment at the lab, with all its in-house celebrations, after-work activities, travelling etc., created a fascinating, never boring joyful working base.

Leaving the Berlin working area and spanning the arc between America and remote outposts in the Carpathians, I would like to name and thank *Gregorio, Camilo, Véro, Sasha, Oleh, Alina, Vlad, Bogdan, and several other's* for their support, helpful discussions, and for feeling home abroad.

Christian, Janusz, Rainer, Anette, Roberto, André, Matze, Jenny, Jörg, and numerous other musicians for providing me with the most efficient and inspiring distraction – making music.

My best friend *Micha* for upholding the connection between heaven and earth while keeping my boots on the ground. Together with his wife *Jule*, son *Tobi* and my wonderful goddaughter *Johanna*, for countless hours of talking, romping, cooking, travelling, and laughing.

My friends *Esco, Andi, Micha R., Neumi, Paule, Darja, Conny & Artur, Ben, Frauke, and Werner* for providing all the valuable entertainment, support, and sympathy.

My parents *Bärbel and Jan*, for their lifelong support. For providing us children with a warm, loving, secure, and open minded environment, complemented by a holistic and decent education. My brothers *Florian and Daniel*, for being great brothers and the just recent support in detailed proofreading.

Last but not least, my marvellous family. *Leo*, for showing me each day what is really important in life and for all the small and big moments of luck. *Magdalena*, just being in the same situation, for our smooth management of everyday life especially on the final meters of our PhD's. For your love and the wonderful life ahead of us.

Abstract

Protected areas are one cornerstone of conservation efforts to safeguard natural habitats from destruction and overexploitation. Still, many of these areas remain less effective than initially envisioned. Besides climate change, main threats originate from enduring human activities. Protected areas are particularly at risk during periods of rapid socio-economic changes, which can trigger widespread land-use change and illegal resource use. The main goal of this thesis is to assess the extent and underlying causes of land-use change in protected areas and forest habitats within the Carpathian Ecoregion. The Romanian Carpathians were selected as a focus area in this study, because they comprise Eastern Europe's largest continuous temperate forest region as well as some of the last and largest tracts of European old-growth forests, and they are a major hotspot of biodiversity. Romania comprises more than half of the Carpathian Ecoregion and it is of particular interest to study the causes and effects of land-use changes, which have emerged after the collapse of socialism in 1989. Post socialist forest cover change was quantified for the last 25 years using Landsat images and an ad hoc developed large area classification technique. Results show widespread forest disturbances, even inside protected areas and old-growth forests. Drivers of these disturbances can be related to institutional change and changes in ownership. The effectiveness of Romania's protected area network in terms of its ability to safeguard biodiversity is most likely decreasing, and intact old-growth forests continue to disappear. This thesis reveals how rapid socio-economic changes may lead to overexploitation, and highlights substantial shortcomings in the effectiveness of protection efforts to safeguard biodiversity and related ecosystem services.

Zusammenfassung

Naturschutzgebiete sind ein essentieller Bestandteil zur Wahrung natürlicher Lebensräume. Oft verfehlt die Einrichtung solcher Schutzzonen jedoch den erwarteten Effekt. Die größte Gefahr liegt hierbei neben dem Klimawandel im direkten Einfluss des Menschen. Besonders in Phasen sozioökonomischen Umschwungs und damit verbundenen Landnutzungsveränderungen oder auch illegaler Nutzung natürlicher Ressourcen, sind Naturschutzgebiete in ihrer Funktion gefährdet. Ziel dieser Arbeit ist somit, Ursachen des Landnutzungswandels sowie dessen Auswirkungen und Ausmaß am Beispiel des rumänischen Teils der Karpaten-Ökoregion abzuleiten. Das Untersuchungsgebiet ist ein wichtiges Zentrum für Biodiversität und in ihm befindet sich Osteuropas größte gemäßigte Waldregion sowie einige der letzten europäischen Urwälder. Rumänien umschließt mehr als die Hälfte der Karpaten und es ist hiernach von besonderem Interesse, Gründe und Auswirkungen des rezenten post-sozialistischen Landschaftswandels zu untersuchen. Mit Hilfe von Landsat Aufnahmen sowie einer ad hoc entwickelten Methode zur Klassifizierung großräumiger Gebiete, wurden Veränderungen in der Waldbedeckung für die post-sozialistische Zeit abgeleitet. Die Ergebnisse offenbaren großflächige Forstveränderungen, auch innerhalb von Naturschutzgebieten und Urwäldern. Institutionelle Umbrüche und eine rapide Umgestaltung in den Eigentumsverhältnissen, wurden als Hauptursachen herausgestellt. Rumänische Naturschutzgebiete erreichen nicht die gewünschte Effektivität und Urwäldern werden weiterhin dezimiert. Die Arbeit verdeutlicht den Einfluss sozioökonomischer Veränderungen auf die Entstehung von Raubbau und legt Defizite in der Effektivität von Schutzbemühungen zum Erhalt der Biodiversität und verbundener Ökosystemleistungen offen.

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Chapter I: Introduction

1 Preface

During the last 50 years, humans have altered the world's ecosystems more intensely and rapidly than in any other period in human history (MA 2005d; Costanza et al. 2007). Since 1945, human population has more than doubled and global economy increased 15-fold, threatening in consequence essential "support systems" for human life (Steffen et al. 2007). An expanding world economy and more and more aggressive globalisation reaches deeper than ever, and even the remotest of places, leading to rapid land-use changes (Lambin et al. 2001). Meanwhile, the magnitude of change seems to fall behind our ability to cope with it (Rull 2011). Scientists across numerous fields argue that mankind has altered our planet in a way that can no longer be measured in Holocene norms. In fact, a new geological epoch may have approached: the *Anthropocene* (Crutzen 2002; Rockstrom et al. 2009; Zalasiewicz et al. 2010). Human driven processes not only change ecosystems, the atmosphere, oceans, or climate, but are responsible for widespread land-use change (Lambin et al. 2001; Costanza et al. 2007), habitat destruction and introduction of invasive species (Bellard et al. 2012). This causes an extensive and irreversible loss in biodiversity (Pereira et al. 2010), leaving anthropogenic traces for millions of years (Underdal 2010; Zalasiewicz et al. 2010). Nonetheless, provided with an especially complaisant environment over the past 12,000 years and by learning how to use it for own needs, humanity was able to progressively develop and thrive (Costanza et al. 2007; Steffen et al. 2011). Building upon this co-evolving human-environmental system, contemporary civilizations, agriculture, villages, cities, industrialisation and global communication, just to name a few, could develop (Steffen et al. 2011).

Yet, many natural systems across the planet are at risk of collapse, the "2010 target" of the *Convention on Biological Diversity* has not been met, and global biodiversity has continued to decline significantly over the past four decades (Butchart et al. 2010; CBD 2010; Hoffmann et al. 2010). Natural habitats diminish, species extinction is accelerating and the Earth-system is pushed closer toward "tipping points", which could result in an unprecedented and dramatic biodiversity loss paired with severe degradations of a wide range of ecosystem services (Rockstrom et al. 2009; CBD 2010; Leadley et al. 2010). For the scientific community it is evident, that a continued, unbound emission of greenhouse gas drives the atmospheric system towards such a crucial "tipping point", causing major social and ecological disruptions that can already be observed today (Roberts 2011).

Furthermore, it is clear that the *Copenhagen Accord* and the *Cancun Agreements* intended to prevent progressive climate change, have failed to pass basic agreements. A new emerging world order resulting from a fundamental hegemonic crisis as well as short-term state and economic interests are seen as the root-causes for this failure (Rull 2011). It thus remains incredibly challenging to manage the trade-offs between immediate human needs at the expense of environmental degradation on the one hand, and maintaining ecosystem goods and services in the long-term on the other (Balmford et al. 2002; Foley et al. 2005). Beyond that, processes of social-environmental change, including climate change and biodiversity loss, are embedded in highly complex systems with large time-lags between cause and effect (Underdal 2010). And this is one of the major problems, since the logic and time scales of politics usually leads to a focus on short-term action dictated by short terms in office. Consequently, a shift from *mitigation* to *adaptation* strategies is happening in the political field (Biesbroek et al. 2010; Underdal 2010) accompanied by an emerging environmental governance (Philip 2011).

Climate change and its impacts are well recognized to affect nearly all terrestrial ecosystems, although its pace and dynamics can vary significantly across the planet (Burrows et al. 2011; Ohlemüller 2011; Bellard et al. 2012). European regions are inevitably affected at large scale, adversely influencing present socio-ecological structures and functions (Biesbroek et al. 2010). Migration pressure increases directly with and seems to be a consequence of environmental changes (IPCC 2007b). Estimations range from 50 to nearly 700 million people migrating by 2050 (Koko 2010). By the same year, the Earth's population may have reached nine billion people which all need to be provided with food (Foley et al. 2011), while at the same time a global warming of an additional 2°C could cause a 20% reduction in wheat harvest yields (Parker 2011). Clearly, land-use will play a key-role in mankind's viability, and it is thus of utmost importance to better understand, evaluate and predict the drivers of land-use decisions and its effects on land-use change (Foley et al. 2005; GLP 2005; Turner et al. 2007).

It is of personal importance - and generally should be every geographer's call - to contribute to a better understanding of a fast changing planet and to help mitigate or, wherever necessary, adapt to an unfavourable development of the coupled human-environmental system. Moreover, he should raise public attention wherever an unsustainable use of Nature's goods and services is discovered.

2 The role of forest ecosystems

Forest ecosystems are essential for human wellbeing, are a refuge for terrestrial biodiversity and an important source for ecosystem services (MA 2005c). Old-growth forests and mountain forests are especially important in sustaining biodiversity (Glatzel 2009; Gibson et al. 2011; Price et al. 2011). Mountain forests, for instance, not only play a crucial role in the provision of fresh water and wood as well as the protection against natural hazards, but are also involved in the physical and mental wellbeing of people (Price et al. 2011). Moreover, forests are the world's largest terrestrial carbon stock and sink (Pan et al. 2011). Nevertheless, due to human-induced deforestation, wildfires, degradation and the accelerating impact of climate change, the fundamental role of forests for biodiversity conservation and the global carbon cycle is at risk (Thompson et al. 2009; Dudley et al. 2010; Price et al. 2011). Climate warming has been shown to be responsible for decreasing the capacity of plants to function as carbon sinks (Zhao and Running 2010) and is now considered one of the most serious future-threat for mountain forests (Glatzel 2009).

A recent satellite-based survey by the United Nations' Food and Agriculture Organization (FAO) revealed that the global forestland is shrinking at an unsustainable rate (FAO 2011). Between 1990 and 2005, 72.9 million ha (net forest loss) — corresponding to a rate of about 10 ha per minute — were lost with rates increasing between 2000 and 2005 (FAO 2011). Negative changes in forest ecosystems are mainly driven by political instability, rapid population growth, pervasive market forces, institutional strengths or weaknesses, and natural- and human-induced disturbances (MA 2005b; Barbier et al. 2010; Andersson and Agrawal 2011). Moreover, the contemporary globalisation fosters the international displacement or leakage of land-use and its role in forest transition¹ (Kastner et al. 2011; Meyfroidt and Lambin 2011). This means that a decreasing availability of productive land on the one hand, and competition with other land-uses in contrast to a growing world

¹ Forest transition: A concept by Alexander Mather describing a long-term land-use change event, whereat a shift from net deforestation to net reforestation takes place at national-scale driven by socio-economic developments (Mather, A.S. (1992). The forest transition. *Area*, 24, 367-379).

population on the other, makes a global forest transition difficult to achieve (Pereira et al. 2010).

Especially in the industrialised countries of northern Europe, land-use changes and conversion of primary forests to managed plantations have almost completely eradicated old-growth forests (Wirth et al. 2009b). But human disturbances within these remaining old-growth forests continue and in many cases have had long-lasting negative effects on species composition and key habitat functions (Frank et al. 2009). Moreover, contrary to the long standing view that old-growth forests are carbon neutral, they continue to sequester carbon for long time periods, but also store more carbon per unit area than any other ecosystem or forest successional stage (Luysaert et al. 2008; Knohl et al. 2009; Wirth 2009; Keeton et al. 2011). Thus, old-growth forests are both an important carbon sink, but also potentially a large carbon source if disturbed (Luysaert et al. 2008). However, despite their ecological importance, old-growth forests are vanishing at an alarming rate (Achard et al. 2009), thereby diminishing the ecosystem services (e.g., genetic resources, protection from natural hazards, riparian functionality) that they provide (Keeton et al. 2007; Wirth et al. 2009a) and threatening the biodiversity they harbour.

3 The role of protected areas for biodiversity conservation

The IPCC recognized adaptive management in forest protected areas as a fundamental factor in biodiversity conservation efforts and the reduction of climate change vulnerability (IPCC 2007a). Protected areas thus not only play a key role in safeguarding existing forests and their ecosystem values, but also in maintaining and enhancing carbon stores (Dudley et al. 2010). Some of these areas, however, are designated only formally (so called “paper parks”) and therefore cannot guarantee effective management. A key point in reducing the loss of biodiversity and dampening the impacts of climate change is thus the expansion and strengthening of protected areas (IPCC 2002; Chape et al. 2005).

Protected areas are furthermore the cornerstone of conservation efforts to safeguard species from destruction and fragmentation of habitat (Myers et al. 2000; Joppa et al. 2008; Cantu-Salazar and Gaston 2010). These threats have been identified as the principal causes of the global biodiversity crisis (Brook et al. 2008; Ehrlich and Pringle 2008; Hoffmann et al. 2010). Moreover, due to their distinct mitigation and adaptation potential, protected areas function as an essential part within the global response to climate change (IPCC 2002). Although some 120,000 designated protected areas exist globally, covering nearly 14% of

the earth surface (Jenkins and Joppa 2009; WDPA 2011), many of them remain less effective than originally envisioned (Gaston et al. 2008; Joppa et al. 2008) and face threats from human activities both within their boundaries and without (Chape et al. 2005; Hansen and DeFries 2007). Therefore, especially given the unprecedented challenge of climate change and human population growth, monitoring the conservation effectiveness of protected areas and understanding their shortcomings are crucial components in the strive for sustaining global biodiversity targets (Chape et al. 2005; McNeely 2008).

Protected areas are also embedded in a complex coupled human-natural system (Liu et al. 2007). There is, however, an increasing awareness that protected areas are not islands, but often are part of a larger ecosystem extending beyond their boundaries (DeFries et al. 2010). What happens outside protected areas clearly influences flows of energy, materials, and organisms in- and out of the area, thus directly affecting the functioning of the ecosystem inside the protected areas (Hansen and DeFries 2007; Jones et al. 2009). This is amplified further by the rapidly intensifying and expanding land-use worldwide, in particular close to and around many of the protected areas (Porter-Bolland 2011). Although protected areas aim at stopping habitat loss inside their boundaries (Bruner et al. 2001; Mas 2005), land-use changes in their surroundings increase their isolation and ultimately reduce the effective size of larger natural ecosystems (DeFries et al. 2005; Newmark 2008; Radeloff et al. 2010). They may also introduce edge effects (Cameron 2006; Hansen and DeFries 2007), and increase extinction debt (Carroll et al. 2004). Assessing and monitoring land-use changes surrounding protected areas across larger ecosystems is therefore a key component in ensuring that protected areas remain effective.

The effectiveness of protected areas is particularly at risk during periods of rapid socio-economic change, such as following wars, revolutions or economic crises (Brechin et al. 2002). They usually trigger and entail widespread land-use change and illegal resource use. Forest loss and degradation are among such extensive changes driven by a combination of political, economic, and institutional factors (MA 2005b). The collapse of socialism in Eastern Europe and the former Soviet Union is a prominent example of such a situation. The transition from command- to market-oriented economies triggered drastic and rapid land-use changes (Ioffe et al. 2004; Kuemmerle et al. 2007; Baumann et al. 2011). At the same time, the infrastructure for nature protection eroded (Wells and Williams 1998), related institutions were weakened, and illegal logging and poaching became widespread in some regions - including protected areas (Soran et al. 2000; Vandergert and Newell 2003; Henry and Douhovnikoff 2008). Together, these are the most prevalent and serious threats

for an effective functioning of protected areas in Europe (Nolte et al. 2010) and elsewhere (Dudley et al. 2010). Moreover, joining the European Union (EU) required these Eastern European countries to substantially enlarge their protected area network (Oszlanyi et al. 2004; Young et al. 2007a), but some of these countries lacked experience in coping with the associated challenges (Gaston et al. 2008). The extent to which such trends have ultimately affected the effectiveness of protected areas in Eastern Europe, however, remains largely unknown.

4 The Carpathian Ecoregion

The Carpathians are Europe's largest mountain range and also its largest continuous temperate forest ecosystem (UNEP 2007) (Figure I-1). Goods and services from temperate forests (such as clean water, wood products and recreation opportunities in relation to the large number of people living in close proximity) make these forests an important ecosystem (Thompson et al. 2009). The temperate forests of the Carpathians are habitat for a high percentage of endemic species and are a key element in the European carbon cycle (Nabuurs et al. 2008; Schulp et al. 2008). Inside, vast natural and semi-natural forests cover an area of about 3000 km² (Anfodillo et al. 2008), including some of Europe's last extensive tracks of high conservation value old-growth forests (Veen et al. 2010). Carpathian forests cover about half of the Carpathian Ecoregion (Ruffini et al. 2006) and are characterized by a patchwork of coniferous, deciduous and mixed stands with a distinct vertical zonation. Main tree species include beech (*Fagus sylvatica*), Norway spruce (*Picea abies*) and silver fir (*Abies alba*). Due to their diversity in plant and animal species, these forests are highly valued for both biodiversity maintenance and nature conservation (UNEP 2007).

Stretching across seven nations, the Carpathians are of outstanding importance for nature conservation because the region has remained relatively undisturbed when compared to Western Europe, is rich in biodiversity, and provides a refuge for large mammals whose populations have been drastically reduced elsewhere in Europe (UNEP 2007; Anfodillo et al. 2008). More than 480 endemic species, threatened mountain species and communities as well as one third (nearly 4000) of European vascular plant species (ANPA 2004) are found in this area. Although divided by political and ethnical frontiers, the Carpathians have a tremendous potential to protect and conserve natural and cultural heritage in a pristine form that is rarely found elsewhere (Oszlanyi et al. 2004). About 16% of the

Carpathians are protected, but this proportion varies widely among countries (Oszlanyi et al. 2004) and the protected area network is still sparse in some regions (Soran et al. 2000; Ioja et al. 2010). Moreover, the enforcement of nature protection laws is often inadequate and corruption fosters illegal resource use (UNEP 2007; Irland 2008; Ioja et al. 2010) with illegal logging having increased substantially (Bouriaud 2005; Kuemmerle et al. 2009).



Figure I-1: Location of the Carpathian mountain range. Source: Author's design based on the following data: SRTM digital elevation model, ESRI Data and Maps Kit, Carpathian Ecoregion Initiative.

Today, enormous pressure is put on the Carpathian ecosystems by changing market forces, new EU transport network, illegal cutting, increasing tourism and new legislations (Buza and Turnock 2004; Maanen et al. 2006). Nevertheless, threats not only arise from anthropogenic impacts, but also from climatic influences (Pullin et al. 2009; Casalegno et al. 2010). Climate change will foster the migration of vegetation zones towards higher altitudes, resulting in an extinction of alpine species (the so-called “bottleneck-trap”), an increase in pests, pathogens and risk of flash floods (UNEP 2007; Price et al. 2011). In a socio-ecological context, climate change is forecast to magnify regional differences with respect to quality and quantity of natural resources and assets (Dudley et al. 2010). In consequence, for the Carpathian Ecoregion and beyond, adaptive management and protection should be focusing on areas reducing effects of climate change (Bellard et al. 2012). This particularly includes forests, because they profoundly contribute to local climate but also serve as climate refuge for biodiversity (Carnaval et al. 2009).

Due to a long history of centralized administration during times of the *Habsburg Empire*, Carpathian forest management has a long record of careful planning (Buza and Turnock 2004). In the communist period, this was somewhat continued with private logging being suppressed. Nevertheless, new challenges and constraints affecting forest cover arose after the fall of the Iron Curtain in 1989. In several Eastern European countries, resulting major socio-economic changes were followed by changes in forest ownership (Nijnik et al. 2009). State-owned forests shifted to private holdings, resulting in a range of different forest uses (MCPFE 2007). The growing number of small-scale forest holdings then slowed down the implementation of sustainable forest management (Turner et al. 1996; Nijnik et al. 2009; Źmihorski et al. 2010). These developments in land ownership regimes consequently triggered land-use transitions (Barbier et al. 2010; Lambin and Meyfroidt 2010) catalyzed by institutional factors, poverty, absence of economic sustainability, population change and people's reaction to new economic opportunities (Lambin et al. 2001). Deforestation (Csóka 2005; Dolisca et al. 2007; Nagendra et al. 2008), reforestation (Southworth and Tucker 2001), and illegal logging (Bouriaud and Niskanen 2003; Kuemmerle et al. 2009) inside and outside protected areas (Nijnik and Van Kooten 2000; Kuemmerle et al. 2007; Knorn et al. in press) are thus the most enduring resulting land-use / land-cover change processes observable.

Overall, unsustainable forest management practices may have affected Carpathian ecosystems and biodiversity in a critical way. Especially in the case of Romania, which encompasses more than half of the Carpathian Ecoregion, land-use changes hamper significantly the safeguarding of biodiversity in the region (Soran et al. 2000). However, it is not only the breakdown of socialism, but also the implementation of EU policies and legislations that had various impacts on land-use and the status of nature conservation (Oszlanyi et al. 2004; Young et al. 2005; Young et al. 2007a). However, to the best knowledge, no research has comprehensively studied and assessed these allocated changes at regional scales throughout Romania.

5 Approach, specific objectives, and method design

Ever since the beginnings of Earth observation, the importance of remotely sensed data to monitor biodiversity has been recognized, and, for several decades now, it has been used to track changes in ecosystem status and distribution (Teder et al. 2007; Muchoney 2008). Moreover, interpreting satellite images is the most accurate and comprehensive approach

for assessing forest cover changes across large areas (Achard et al. 2009; FAO 2011). Since forest cover is correlated with species habitat and carbon storage, forest disturbance is an indirect indicator for protected area effectiveness (Joppa and Pfaff 2010).

Assessing the effectiveness of protected areas is a challenging task. Statistical data is often outdated, unavailable or incomparable between countries. Spatial data on land-cover (such as national maps) vary, if at all existent, in quality, data relevance, mapping algorithms, and are often restricted to specific user groups. The use of remote sensing satellite images is thus a promising and invaluable alternative, since they are widely used to assess rates and spatial patterns of land-use and land-cover changes as well as status and trends of biodiversity and ecosystems (Lambin and Geist 2006; Muchoney 2008; Jones et al. 2009). Without doubt, their application in an environmental context can contribute to mitigate negative ecological impacts (Jones et al. 2009). Satellite remote sensing became a critical and universal tool for natural resource managers, providing consistent measurements on a landscape scale and detecting both slow trends over time as well as abrupt changes of land-cover (Kennedy et al. 2009). *Landsat* satellite data is the most widely used data type for land-cover mapping thanks to its 35-year data record and its relatively high spatial resolution (Cohen and Goward 2004; Wulder et al. 2008). This wealth of data offers great opportunities to determine changes in forest cover consistently across space and time (Hansen et al. 2010; Zhao and Running 2010; FAO 2011). Moreover, using *Landsat* data based disturbance dynamics enables to draw conclusions about the effectiveness of protected areas (Young et al. 2006; Fraser et al. 2009; Huang et al. 2009). *Landsat's* parameters with a swath width of 185 km, 30 m by 30 m pixel size and a 16-day repeat cycle are suitable prerequisites for land-cover mapping on a landscape scale (Cohen and Goward 2004). Furthermore, the USGS's decision to provide free access to all *Landsat* data holdings now offers great opportunities for comprehensive yet cost efficient land-cover classifications.

Assessing forest cover and forest cover changes is a widely respected and robust indicator of environmental integrity (Muchoney 2008; Zimmerer 2009). Compared to other sensors such as *MODIS* (*Moderate Resolution Imaging Spectroradiometer*), *Landsat* satellite images provide sufficiently high spatial detail to classify and separate forest from other land-cover classes (Cohen et al. 1996). Using satellite-based forest cover maps, this approach thus enables objective large-scale estimations of the effectiveness of protected area management (Bruner et al. 2001; Brechin et al. 2002; Butchart et al. 2010), independent of official forest management agencies. Moreover, outcomes are immediately

available to the research community, conservation planning and for potential management implications (Potapov et al. 2011).

The core contribution of the present work is to investigate large area landscape dynamics across the main part of the Carpathian Ecoregion, to relate those to underlying socio-economic driving forces, and to assess the effectiveness of the Carpathian protected area network. A methodological prerequisite for this work included the development of a novel technique to classify land-cover across large areas. Chapter II describes this so-called “chain classifications” technique (i.e., the classification of *Landsat* images based on the information in the overlapping areas of neighboring scenes), which was designed using *Landsat* data that covers a representative part of the Carpathians. The *chain classification* approach was then not only successfully applied within the framework of this present study, but also beyond: First, within a forest cover change analysis focusing on the Ukrainian part of the Carpathians and revealing widespread illegal logging (Appendix A). Second, within a regional analysis focusing on the entire territory of Romania and assessing the impact of forest restitution on the terrestrial carbon balance (Appendix B). The following Chapter III analyses the effectiveness of protected areas for the northern part of the Romanian Carpathians based on multi-temporal change classifications using *Support Vector Machines* (SVM). In particular, forest cover changes inside and outside protected areas were derived and associated with underlying socio-economic as well as institutional drivers. In the last core chapter (Chapter IV), a comprehensive analysis on the status of old-growth forests in the Romanian Carpathians and the effectiveness of their protection is discussed.

The main objectives and contributions of this dissertation are the following:

1. The development of a simple, robust, and reproducible method for large area land-cover classification with minimal requirements on image pre-processing and training data;
2. An assessment of the effectiveness of selected Romanian protected areas at preventing unsanctioned logging and an investigation of the effects of forest restitution on logging rates and patterns;
3. An analysis of the extent of old-growth forest disturbances in Romania and the effectiveness of protected areas to safeguard these forests.

6 Structure of this thesis

As described above, this thesis is build around the three main sections (Chapters II-IV), each relating to one of the research objectives discussed above. Chapter V then presents a synthesis of the outcomes of the individual chapters, summarising their findings, drawing more general conclusions regarding their implications and discussing future directions. Chapters II-IV make up the core of this thesis and were written as stand-alone manuscripts to be published in internationally recognized, peer-reviewed journals. They thus fulfil the formal requirements of a cumulative doctoral dissertation. Since certain repetitions are inherent in the nature of a cumulative dissertation (such as sections discussing background, study area, methods, results, conclusions, etc.), a certain amount of recurrence in the thesis is unavoidable. The three core chapters were published or submitted as follows:

Chapter II: Knorn, J., Rabe, A., Radeloff, V.C., Kuemmerle, T., Kozak, J., & Hostert, P. (2009). Land cover mapping of large areas using chain classification of neighboring Landsat satellite images. *Remote Sensing of Environment*, 113, 957-964.

Chapter III: Knorn, J., Kuemmerle, T., Szabo, A., Mindrescu, M., Keeton, W.S., Radeloff, V.C., Abrudan, I.V., Griffiths, P., Gancz, V., & Hostert, P. (2012). Forest restitution and protected area effectiveness in post-socialist Romania. *Biological Conservation*, in press.

Chapter IV: Knorn, J., Kuemmerle, T., Radeloff, V.C., Keeton, W.S., Griffiths, P., Hagatis, A., Gancz, V., Biriş, I.A., & Hostert, P. (2012). Continued loss of temperate old-growth forests in the Romanian Carpathians despite an increasing protected area network. *Environmental Conservation*, submitted.

Two appendices supplement the thesis. Both present the successful implementation of the method developed under the framework of Chapter II and underpin the findings of Chapters III and IV. Both appendices were co-authored as independent pieces of research for publication in peer-reviewed journals. The references for the appendices are:

Appendix A: Kuemmerle, T., Chaskovskyy, O., Knorn, J., Radeloff, V.C., Kruhlov, I., Keeton, W.S., & Hostert, P. (2009). Forest cover change and illegal

logging in the Ukrainian Carpathians in the transition period from 1988 to 2007. *Remote Sensing of Environment*, 113, 1194-1207.

Appendix B: Olofsson, P., Kuemmerle, T., Griffiths, P., Knorn, J., Baccini, A., Gancz, V., Blujdea, V., Houghton, R.A., Abrudan, I.V., & Woodcock, C.E. (2011). Carbon implications of forest restitution in post-socialist Romania. *Environmental Research Letters*, 6, 045202.

**Chapter II:
Land-cover mapping of large areas using chain
classification of neighboring Landsat satellite
images**

Remote Sensing of Environment 113 (2009) 975-964

Jan Knorn, Andreas Rabe, Volker C. Radeloff, Tobias Kuemmerle,
Jacek Kozak, and Patrick Hostert

Abstract

Satellite imagery is the major data source for regional to global land-cover maps. However, land-cover mapping of large areas with medium-resolution imagery is costly and often constrained by the lack of good training and validation data. Our goal was to overcome these limitations, and to test chain classifications, i.e., the classification of Landsat images based on the information in the overlapping areas of neighboring scenes. The basic idea was to classify one Landsat scene first where good ground truth data is available, and then to classify the neighboring Landsat scene using the land-cover classification of the first scene in the overlap area as training data. We tested chain classification for a forest/non-forest classification in the Carpathian Mountains on one horizontal chain of six Landsat scenes, and two vertical chains of two Landsat scenes each. We collected extensive training data from Quickbird imagery for classifying radiometrically uncorrected data with Support Vector Machines (SVMs). The SVMs classified 8 scenes with overall accuracies between 92.1% and 98.9% (average of 96.3%). Accuracy loss when automatically classifying neighboring scenes with chain classification was 1.9% on average. Even a chain of six images resulted only in an accuracy loss of 5.1% for the last image compared to a reference classification from independent training data for the last image. Chain classification thus performed well, but we note that chain classification can only be applied when land-cover classes are well represented in the overlap area of neighboring Landsat scenes. As long as this constraint is met though, chain classification is a powerful approach for large area land-cover classifications, especially in areas of varying training data availability.

1 Introduction

Large area land-cover maps derived from satellite images play a key role in global, regional and national land-cover and land-use assessments, carried out for example by the United Nations (UN), the Food and Agricultural Organization (FAO), or the United States Geological Survey (USGS) (Cihlar 2000; Vogelmann et al. 2001; Franklin and Wulder 2002; Homer et al. 2004). Such classifications allow assessments of broad-scale forest fragmentation (Riitters et al. 2002), carbon sequestration potential (Cruickshank et al. 2000; Niu and Duiker 2006), or the Wildland Urban Interface (Radeloff et al. 2005). Therefore, large area land-cover classifications present a basic prerequisite for many scientific applications (Wulder et al. 2008).

Landsat satellite data is the most widely used data type for land-cover mapping because of its 35-year data record and its relatively high spatial resolution (Cohen and Goward 2004; Wulder et al. 2008). Landsat data will become even more valuable as the Landsat Data Continuity Mission (NASA 2008; Wulder et al. 2008) ensures future data availability. Decreasing costs, the availability of free Landsat data in the Geocover dataset (Tucker et al. 2004), the “Mid-decadal Global Land Survey” (Morris et al. 2008) and the USGS’ decision to provide free access to all Landsat data holdings offer opportunities for large area land-cover classifications using Landsat imagery.

Unfortunately, Landsat image classifications are commonly conducted on one scene at a time, which limits the rapid analysis of large areas (Cihlar et al. 1998; Cihlar 2000) and requires that adequate ground truth data are available for each scene. For large area classifications, three approaches have been proposed and tested before: single scene classification and subsequent mosaicking, mosaicking of images and subsequent classification of the image mosaic as a whole (Cihlar 2000), and signature extension. In signature extension, a classifier is trained on one scene and the resulting signatures are applied to different scenes in space or time (Pax-Lenney et al. 2001). Signature extension is promising, but has to account for differences in topography, phenology, illumination, landscape variability, and atmosphere that result in spectral differences among images. Tests in northwest Oregon showed that accuracy declined by 8-13% (depending on the atmospheric correction method applied) when extending the classifier from an initial training image across space to nearby scenes (Pax-Lenney et al. 2001). Across northern

Canada, classification accuracy dropped approximately by 50% when using signature extension for images that were about 1,500 km apart (Olthof et al. 2005).

A promising approach for mosaicking images prior to classification is ‘applied radiometric normalization’ (Cohen et al. 2001). Here, the overlap area between neighboring Landsat images is used to extend information gained from a source image to neighboring images, thereby creating a seamless mosaic for the classification. The first step is to develop a relationship between the spectral measurements in the source image, and continuous forest variables, such as percent vegetation cover or stand age that are available from ground truth data (Cohen et al. 2001). The second step is to apply the regression equations that were developed, and predict the forest structure attributes across the entire source image. In the third step, the map with the predictions in the overlap area is used as ground truth to develop new regression equations for the neighboring image, which has most likely different phenology and atmospheric conditions. In the fourth step, these regression equations are then applied to the entire neighboring image. The resulting map of continuous forest structure attributes for the entire study area can then be classified into different forest types. When testing this approach in a 73,000 km² study area in western Oregon based on two Landsat TM source images, estimates for four forest cover attributes resulted in an overall accuracy of 66% (Cohen et al. 2001).

Signature extension and the mosaicking of images prior to classification have great potential for classifying large areas using Landsat imagery. However, they require considerable effort to match multiple images radiometrically. Here, we propose a new approach to large area land-cover classification that fills a gap between single scene classification on one hand and signature extension or mosaicking on the other hand. We suggest the term ‘chain classification’ for this method.

Chain classification is similar to applied radiometric normalization in that it uses the overlap area among neighboring Landsat scenes, but we propose classifying one initial scene and then using the classification in the overlap area to train a classifier for a neighboring image. Once the second image is classified, it can be used as a new initial scene to classify a third image and so forth. One potential advantage of chain classification is that it does not require atmospheric correction or regression matching of scenes to account for radiometric differences. It can be applied both in horizontal directions (across track), and in vertical direction (along track). Furthermore, large area land-cover maps often cover several countries or different land ownership regimes. The availability of

spatially well distributed training and validation data is often limited in such situations. Chain classification may offer a solution to this problem by using the image with the best available ground truth data as the starting image in the image chain, and by providing training data for neighboring images from the image chain itself.

In principal, any classification algorithm could be used for chain classification. However, Support Vector Machines (SVMs), a fairly recently developed non-statistical classifier based on machine learning theory (Vapnik 1999) offer some method-inherent advantages. Comparisons with other classification algorithms show that SVMs outperform or are at least as accurate as other parametric or non-parametric classifiers (Huang et al. 2002; Pal and Mather 2005; Dixon and Candade 2008).

SVMs are able to separate complex classes (Melgani and Bruzzone 2004) such as in forest change analysis (Huang et al. 2008). In the SVM, the location of decision boundaries for optimal class separation is determined using kernel functions representing non-linear decision surfaces (Vapnik 1995; Pal and Mather 2005). By constructing the optimum hyperplane in feature space between two classes, an SVM is a binary classifier focusing on the classes of interest only. To determine this hyperplane, only the edges between the class distributions are described based on a relatively small amount of training data (Foody and Mathur 2004; Foody et al. 2007; Mathur and Foody 2008).

In summary, the overarching goal of this study was to develop a simple, robust, and reproducible method for large area land-cover classification with minimal requirements for image pre-processing and training data. To do so, we tested chain classification of forest and non-forest based on the overlapping areas between Landsat scenes in the Carpathian Mountains in Eastern Europe.

2 Data and methods

2.1 Study area

We selected the Carpathians as a study area to test chain classification. The Carpathians represent a fairly homogeneous ecoregion with mostly similar environmental conditions. However, the study area includes seven countries with significant differences in forest type, non-forest land-cover classes, geology, and land-use patterns, and exhibits elevation-dependent vegetation gradients. This variability generates an interesting test case to investigate the feasibility of chain classification.

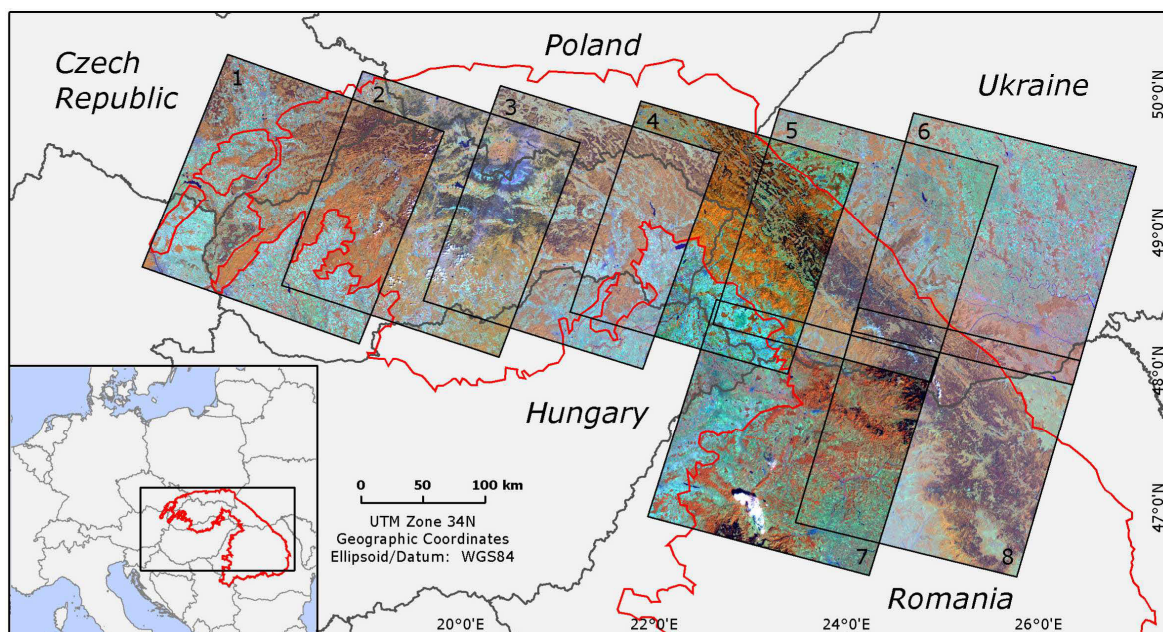


Figure II-1: The Carpathian mountain range and Landsat scenes in R-G-B bands 4-5-3 (border of the Carpathian Ecoregion overlaid in red). Numbering of Landsat scenes corresponds to numbering in the text (sources: GLCF, ESRI Data, Carpathian Ecoregion Initiative).

The Carpathians are located in central Europe and include parts of Czech Republic, Slovakia, Poland, Ukraine, Hungary, and Romania (Webster et al. 2001) (Figure II-1). The study area covers about 185,000 km². The climate of the Carpathians is temperate-continental. Geology varies from Carpathian flysch, consisting of sandstone and shale layers, sedimentary rocks (mainly limestone), to a variety of crystalline rocks. Elevations range from 300 m to over 2000 m (above sea level) in the alpine belt of the Tatra Mountains and the Southern Carpathians (UNEP 2007).

The forests of the study area are a patchwork of deciduous, coniferous, and mixed stands, with pronounced vegetation zones along the elevation gradient (UNEP 2007). Mixed deciduous forests, dominated by pedunculate oak (*Quercus robur*), lime (*Tilia cordata*) and hornbeam (*Carpinus betulus*), dominate the foothill zone. European beech (*Fagus sylvatica*), silver fir (*Abies alba*), Norway spruce (*Picea abies*) and sycamore (*Acer pseudoplatanus*) are typically found in the montane zone (Perzanowski and Szwagrzyk 2001). In some places, the montane zone is almost solely covered by conifers, especially spruce plantations. At the timberline (~1500 m), stone pine (*Pinus cembra*) stands exist (UNEP 2007). Overall, about 60% of the Carpathian Ecoregion is covered by forest (UNEP 2007). A history of intense land-use affected most forests, transforming the landscape into a complex pattern of forests, arable land, and pastures, varying significantly between countries and regions (Turnock 2002; Kuemmerle et al. 2006; Kozak et al. 2008). In

particular, the foothill zones and plains are dominated by agricultural land-use and forests are only small and scattered.

2.2 Satellite data and pre-processing

We used the optical bands of 9 Landsat Enhanced Thematic Mapper Plus (ETM+) images recorded between 2000 and 2002 to test chain classification (Table II-1). Eight images were provided by the University of Maryland Global Land Cover Facility (GLCF), and one Level 1G scene (186/26) was purchased because of cloud coverage in the GLCF data. A post-processed digital elevation model (DEM) from the Shuttle Radar Topography Mission (SRTM) was acquired from the GeoPortal provided by the Consortium for Spatial Information within the Consultative Group on International Agricultural Research (CGIAR-CSI 2004; Reuter et al. 2007) and resampled to 30 m to match the resolution of the Landsat images. We orthorectified the additional 186/26 image using space resection based collinearity equations. The corresponding GLCF image of 2000 served as a base map for automatic image-matching. 359 evenly distributed ground control points (GCPs) with an overall root-mean-squared-error (RMSE) of <0.5 were selected using an Automatic Point Measurement software tool (Leica Geosystems 2006). The image was rectified to Universal Transverse Mercator (UTM) zone 34 and the World Geodetic System (WGS) 84 datum and ellipsoid. The images 189/026, 184/026 and 184/27 were reprojected to UTM zone 34. We resampled all images to 30 m resolution using nearest neighbor resampling to ensure consistency among images. For the GLCF images, the RMSE-based geodetic accuracy is < 0.5 pixels (Tucker et al. 2004). We did not screen for haze or disturbance factors other than clouds and no radiometric correction was applied. Clouds and cloud shadows were digitized and masked out for the analysis.

Table II-1: Landsat images used in this study.

Id	Path/row	Acquisition date
1	189/26	08/02/2000
2	188/26	05/26/2001
3	187/26	08/20/2000
4	186/26	06/10/2000
5	185/26	06/03/2000
6	184/26	08/21/2002
7	185/27	08/22/2000
8	184/27	07/04/2002

2.3 Image classification with Support Vector Machines

SVM classification is based on delineating two classes by fitting an optimal separating hyperplane to the training samples. The hyperplane is constructed by maximizing the margin between class boundaries and is described by a subset of the training samples, the so-called 'support vectors' (Boser et al. 1992; Cortes and Vapnik 1995; Foody et al. 2007). SVMs need training data that optimize the separation of the classes rather than describing the classes themselves (Foody and Mathur 2006).

Using a radial basis function, class distributions with non-linear boundaries can be mapped into a high dimensional space for linear separation (Huang et al. 2002). Training the SVM with a Gaussian radial basis function requires setting two parameters: C is a regularization parameter that controls the trade-off between maximizing the margin and minimizing the training error, while γ describes the kernel width. A small C -value tends to emphasize the margin while ignoring the outliers in the training data, while a large C -value may overfit the training data. A comprehensive description of SVMs can be found in Burges (1998) and Cristianini and Shawe-Taylor (2000). Detailed introductions in a remote sensing context are provided in Huang et al. (2002), and Foody and Mathur (2004). Training, classification, and accuracy assessment were carried out using imageSVM (Janz et al. 2007), an IDL/ENVI based tool for SVM classification of remote sensing images using the LIBSVM version 2.84 (Chang and Lin 2001).

2.4 Training and validation data

Training and validation data, here referred to as 'reference data', was collected using Quickbird images in Google Earth™ (<http://earth.google.com>). About 160 Quickbird images acquired between 2002 and 2007 were available in Google Earth™ (Figure II-2) covering approximately 24% of our study area.

Reference data were collected using a random sampling design (Wang et al. 2005; Lee and Huang 2007). A random sample of 1400 reference points per Landsat image was selected within the area covered by Quickbird imagery in each scene. We chose this number of points, after initial tests based on learning curves showed that selecting more than 500 points per class did not improve classification accuracy significantly. Points were visually classified as either forest or non-forest. The forest class in our study refers to forest as land-cover and includes primary forests as well as plantations, all forest types in the study area (deciduous, mixed, coniferous forests) and all age classes. All other land-cover types (e.g.

settlements, cropland, pastures, and water) were defined as the non-forest class. Because SVMs can delineate multi-modal classes in feature space, we did not have to separate the non-forest training data into individual land-cover classes. All reference points were also cross-checked visually on the Landsat images to account for changes that occurred between the acquisition dates of Landsat and Quickbird images. Points indistinct in Quickbird or covered by clouds were rejected from the analysis. At most, 3% of the random samples were removed.

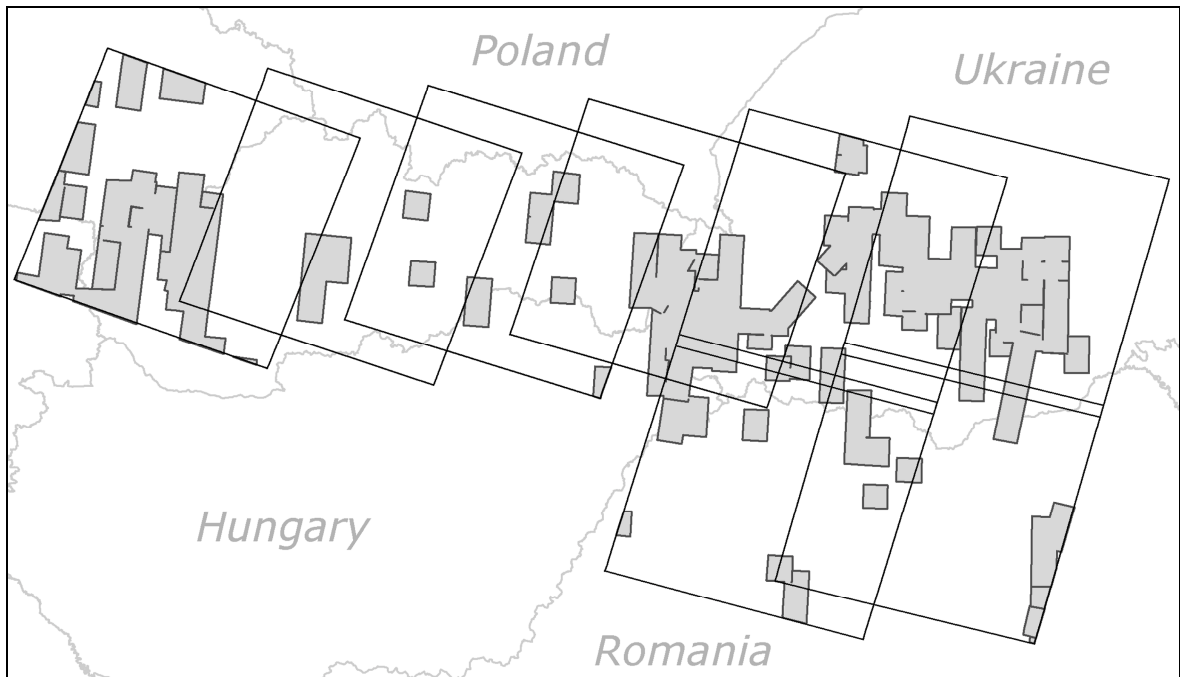


Figure II-2: Distribution of high resolution Quickbird data (grey polygons) from Google Earth™.

In a first step, all scenes were classified individually based on the reference data that had been collected in each scene. We used these classifications as the benchmark against which we compared the chain classification results, and refer to the individual and independent classifications as 'reference classifications' (Figure II-3 top). We used cross-validation to obtain a robust estimate of the accuracy of these reference classifications (Steele 2005). Using ten-fold cross-validation, we split all available ground truth points into training (90%) and validation (10%) samples and then classified each image 10 times for all 10 possible splits. Based on each classification, an error matrix, overall accuracy, user's and producer's accuracy, and kappa were calculated (Congalton 1991; Foody 2002). The derived accuracy measures for each classification were then averaged to calculate mean error estimates (Friedl and Brodley 1997). The final classification was based on an SVM trained with 100% of the ground truth data, and the mean error estimate is thus a conservative estimator of the true accuracy (Burman 1989). Overall accuracies based on

the ten-fold cross-validation ranged from 92.10% for scene 2 to 98.93% for scene 6, with an average of 96.26% (Table II-2).

2.5 Chain classifications using SVMs

The first step in the chain classification was to identify the overlapping area between a reference classification serving as an initial scene and the neighboring scene to be classified (Figure II-4). Within the overlap area, 500 training points each for forest and non-forest were randomly selected. We chose 500 training points, after initial tests based on learning curves showed that selecting more training points in the overlap area did not improve classification accuracy. All training points were at least two pixels apart from forest/non-forest boundaries to account for geometric uncertainty between GLCF scenes (Tucker et al. 2004). The training data from the overlap area formed the input for the SVM classification of the neighboring target scene (Figure II-3 bottom). This procedure was repeated along a chain of overlapping, neighboring scenes. Each former neighboring scene served as a new initial scene in the next step along the chain.

Across track chain classification - Across track chain classification examined Landsat image neighbors in East-West direction. A total of 10 different chain classifications were possible among the six images in the northern row (Figure II-1). At the latitude of our study area, the size of the overlap area between two Landsat scenes across track was about 12,000 km², equaling 35% of a scene.

We also classified chains of more than one neighboring scene. The longest chain included five scenes that were classified based on one initial scene (Figure II-1). We expected decreasing chain classification accuracy with increasing chain length.

Along track chain classification - Four tests were possible to test chain classification along track (Figure II-1). The average along track overlap area was 3800 km², equaling 11% of the scene area. We expected that classification accuracy would be lower compared to across-track chains, given this rather small overlap area.

Across track chain classification based on two classified initial scenes - Last but not least, we tested if chain classification results would improve if the middle image in a chain of three images was classified based on training data from the two images at the ends of the chain. We expected that chain classification would perform better for the centered scene, because not only one but both overlap areas were used for chain classification.

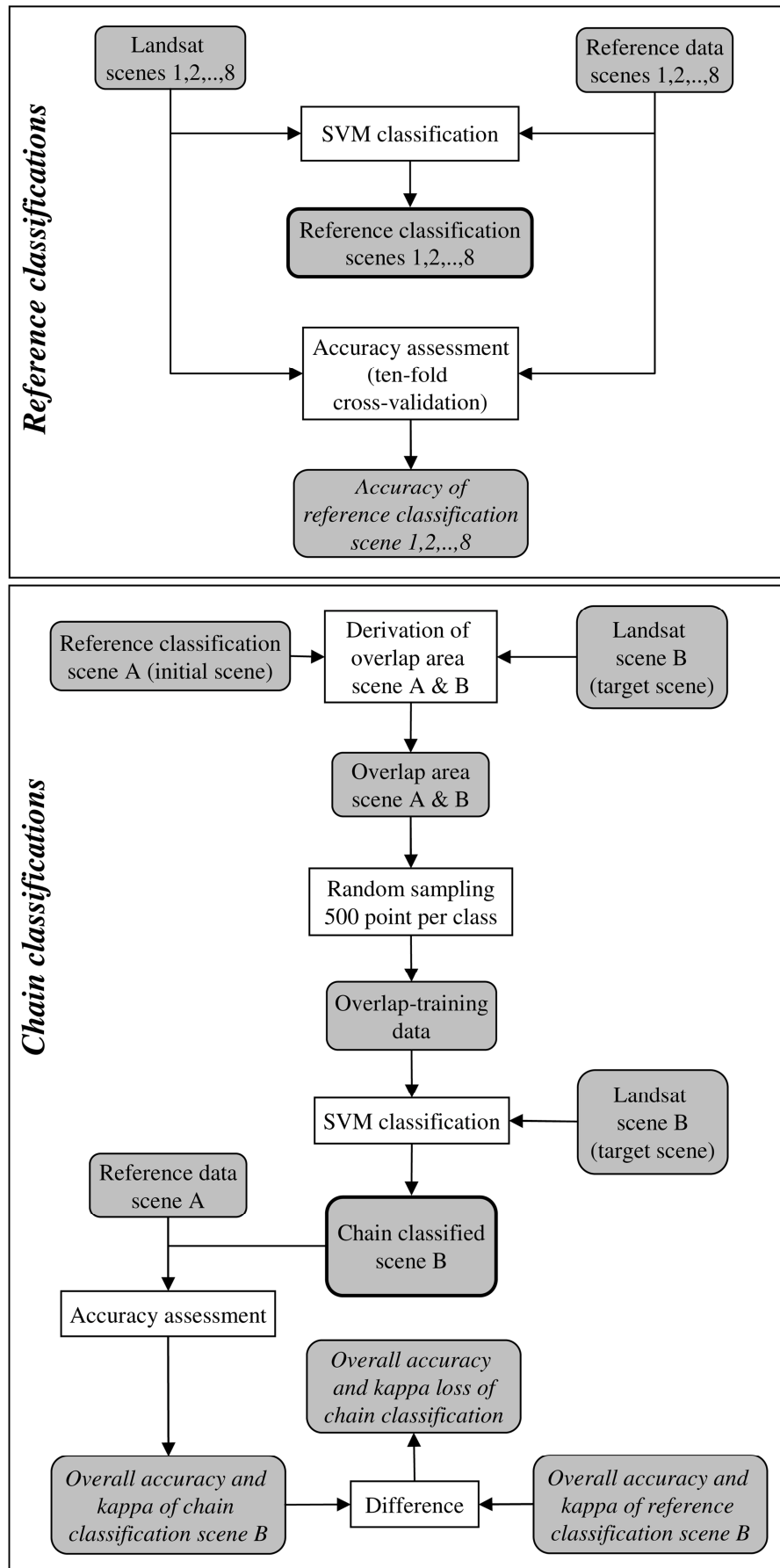


Figure II-3: Processing scheme for chain classifications. Top: Derivation of the reference classifications. Bottom: Chain classification procedure.

Table II-2: Accuracy assessment for reference classifications.

Id	Overall (%) accuracy	Kappa value	User's accuracy (%)		Producer's accuracy	
			Forest	Non-Forest	Forest	Non-Forest
1	96.76	0.926	95.74	97.28	94.41	97.92
2	92.10	0.842	91.76	92.68	93.24	90.87
3	97.48	0.950	97.86	97.13	97.14	97.83
4	97.04	0.938	95.40	98.17	97.18	96.94
5	96.34	0.927	95.89	96.90	96.82	95.86
6	98.93	0.977	98.34	99.28	98.68	99.07
7	95.20	0.886	90.96	97.22	93.41	95.96
8	96.23	0.918	96.14	96.59	98.04	93.15

The accuracy of the chain classifications was assessed independently for each scene, using the available reference data for the respective target scene. Ultimately though, the absolute accuracy was less important to us than the loss in accuracy that was caused when classifying a scene using chain classification. The performance of chain classification itself was assessed calculating the overall accuracy loss and kappa loss between a chain-classified target scene and its respective reference classification (Figure II-3 bottom). For example, in test 1-2-3-4 (Table II-3, test C), scene 1 was the initial scene, scenes 2 and 3 were intermediate chain classifications, and scene 4 was the chain-classified target scene. The overall accuracy loss and kappa loss were calculated by comparing the accuracy of the chain-classified scene 4 and the reference classification of scene 4.

Table II-3: Results of across track chain classification. A.L. = accuracy loss (%) of chain-classified target scene compared to the respective reference classification; K.L. = kappa value loss of chain-classified target scene compared to the respective reference classification. Target scenes in bold.

Test A	A.L.	K.L.	Test B	A.L.	K.L.	Test C	A.L.	K.L.	Test D	A.L.	K.L.	Test E	A.L.	K.L.
1-2	4.50	0.088												
2-3	1.05	0.021	1-2-3	2.41	0.049									
3-4	1.43	0.032	2-3-4	2.38	0.051	1-2-3-4	2.08	0.044						
4-5	0.71	0.017	3-4-5	4.10	0.085	2-3-4-5	4.45	0.092	1-2-3-4-5	3.07	0.065			
5-6	0.26	0.009	4-5-6	0.53	0.015	3-4-5-6	0.26	0.009	2-3-4-5-6	0.66	0.018	1-2-3-4-5-6	0.82	0.022
6-5	2.95	0.062												
5-4	2.33	0.051	6-5-4	6.07	0.137									
4-3	0.84	0.017	5-4-3	1.62	0.033	6-5-4-3	3.55	0.071						
3-2	3.24	0.063	4-3-2	3.10	0.060	5-4-3-2	3.59	0.037	6-5-4-3-2	13.20	0.258			
2-1	1.82	0.046	3-2-1	2.27	0.057	4-3-2-1	0.88	0.024	5-4-3-2-1	2.10	0.053	6-5-4-3-2-1	9.39	0.240
Mean	1.91	0.041	Mean	2.81	0.061	Mean	2.47	0.034	Mean	4.76	0.098	Mean	5.11	0.131

Additionally, we calculated the pixel-wise agreement between two individually derived chain classifications for the same target scene to indirectly evaluate the chain classification performance. If, for example, scene 2 was the target of a chain classification that started from scene 1 and from scene 3, respectively, pixel-wise agreement was calculated as the agreement of the two resulting scene 2 classifications.

3 Results

3.1 Across track chain classification

The results for chain-classifying the direct neighbors had overall accuracy losses ranging from 0.26% for scene 5 to 6 (kappa loss 0.0091) to 4.50% for scene 1 to 2 (kappa loss 0.0882) with an average of 1.91% (kappa loss 0.0408) (Table II-3, test A; Figure II-5). Both tests including scene 2 as a target scene, resulted in the highest overall accuracy and kappa losses.

Accuracy and kappa loss tend to increase as more scenes were added to a classification chain (Table II-3). Average overall accuracy loss ranged from 1.91% (kappa loss 0.0408) for two scenes in a chain up to 5.11% (kappa loss 0.1307) for six scenes in a chain.

The best pixel-wise overall agreement between two different chain classifications for one scene from either neighbor was achieved for scene 3 (95%, Table II-4, test A). The chain classification of scene 2, on the other hand, exhibited only 84.95% in agreement. Average pixel-wise agreement was 91.53% (Table II-4). The same tests, but with three scenes in a chain resulted in pixel-wise agreements of 95.94% with scene 3 as a target scene, and 88.80% with scene 4 as a target scene (Table II-4, test B). Resulting average agreement was 92.37%.

3.2 Along track chain classification

Contrary to our expectation, along track chain classification outperformed across track chain classification. Chain classification using only the small portion of overlap between scenes in North-South direction had an average overall accuracy loss of 1.60% (kappa loss 0.0362) (Table II-5). Highest overall accuracy loss occurred using chain classification from scene 6 to 8 with 3.76% (kappa loss 0.0808) and lowest from scene 8 to 6 with 0.33% (kappa loss 0.0105).

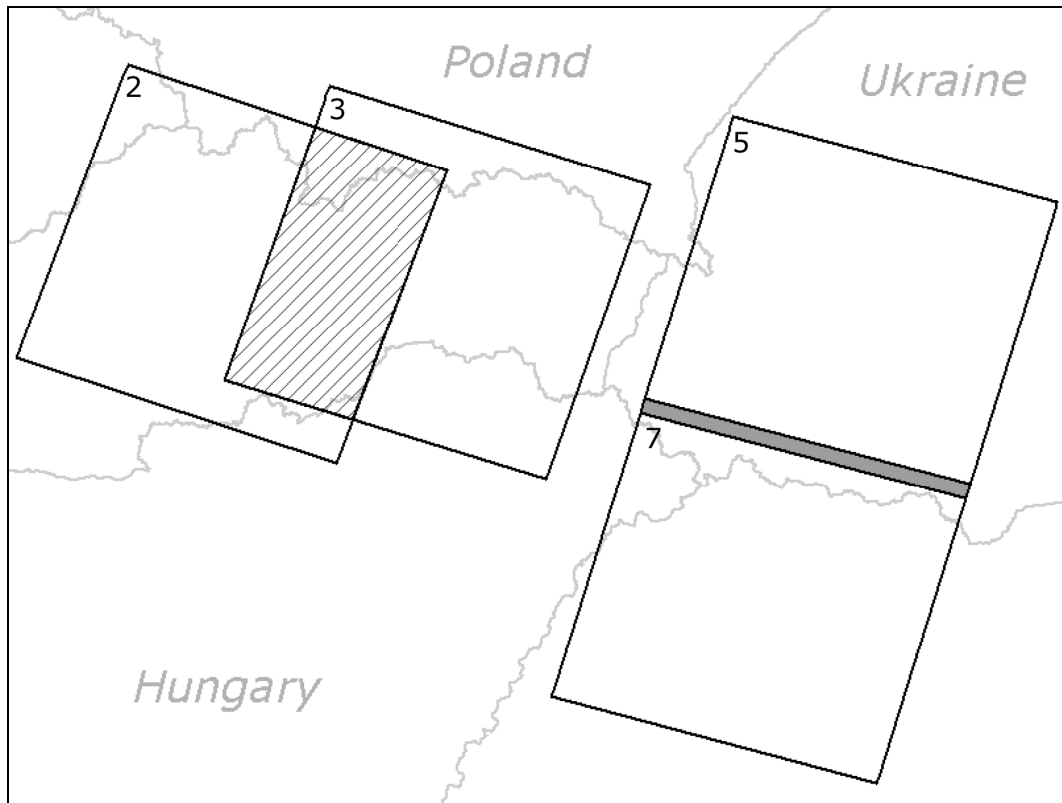


Figure II-4: Neighboring scenes with respective overlap areas. Striped: overlap area between two scenes across track; grey: overlap areas between two scenes along track.

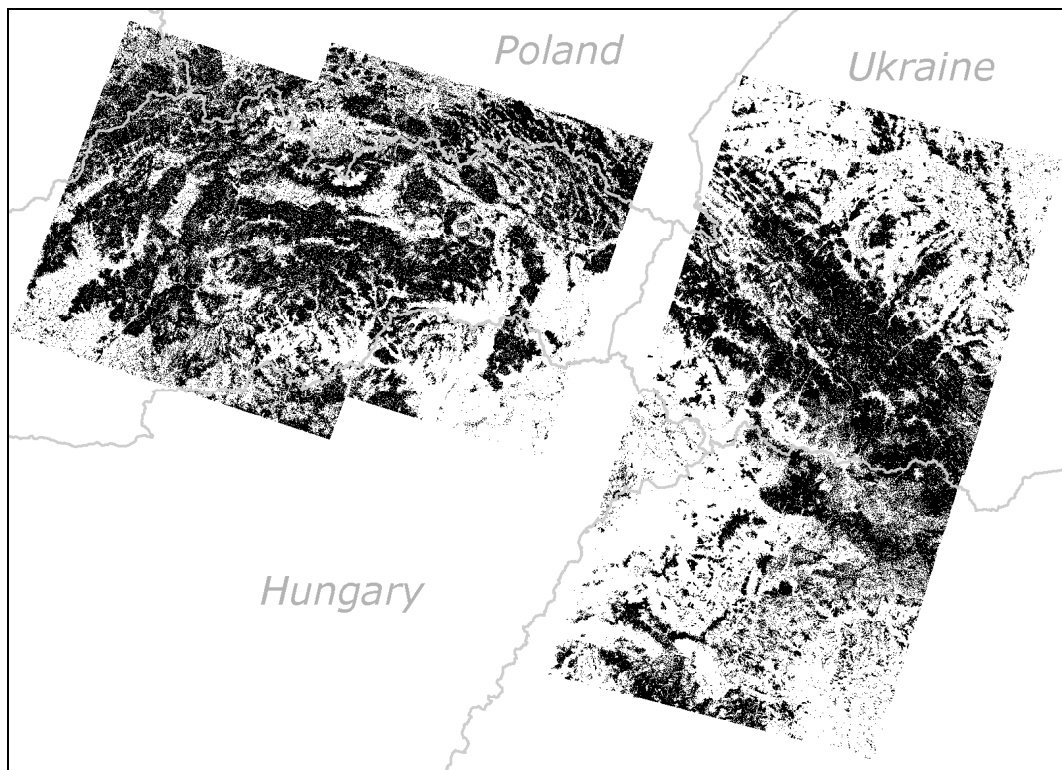


Figure II-5: Results of two chain classifications (forest in black). Scenes 2 and 5 classified initial scenes; scenes 3 and 7 chain-classified target scenes.

Table II-4: Overall agreement (O.Ag.) (%) between two individually derived chain classifications of the same target scene (bold) across-track.

Test A	O.Ag.	Test B	O.Ag.
1-2	84.95	1-2- 3	95.94
3-2		5-4- 3	
2-3	95.00	2-3- 4	88.80
4-3		6-5- 4	
3-4	92.88		
5-4			
4-5	93.30		
6-5			
Mean	91.53	Mean	92.37

Table II-5: Accuracy- and kappa-losses along track.

Test	A.L.	K.L.
5-7	0.36	0.0107
7-5	1.98	0.0427
6-8	3.37	0.0808
8-6	0.33	0.0105
Mean	1.60	0.0362

Table II-6: Results of across track chain classification based on two classified initial scenes.

Test	A.L.	K.L.
1-2- 3 -4-5	2.12	0.0429
2-3- 4 -5-6	0.78	0.0189
Mean	1.45	0.0309

3.3 Across track chain classification based on two classified initial scenes

Chain classifications of a target scene based on its two overlap areas with neighboring scenes did not result in notably higher classification accuracies (Table II-5). A test based on scene 3, for example, results in an overall accuracy loss of 2.12% (kappa loss 0.0429), while for scene 4 the overall accuracy loss is only 0.78% (kappa loss 0.0189).

4 Discussion

We tested ‘chain classification’, a new approach to classify land-cover for large areas that uses a classification in one image to train a classifier for a neighboring image. The average loss of accuracy when comparing *across track chain classifications* to reference classifications was only 1.91% with two images in a chain, 2.81% with three, 2.47% with four, 4.76% with five and 5.11% with six scenes, respectively. Average pixel-wise agreement between two individually derived chain classifications of the same scene across

track was 91.53% for two scenes and 92.37% for three scenes in a chain. These outcomes highlight that chain classification is a robust way to map land-cover across several scenes. Chain classification across track works well with up to four scenes in a chain in our case (Table II-3, tests A to C). Outliers in these tests followed no decisive pattern and are likely due to irregularly distributed reference data (Figure II-2).

The small average loss of accuracy of *along track chain classification* suggests that our approach works also well in North-South direction even though the overlap area is fairly small in this case. The variance within the accuracy losses showed again no clear pattern.

Across track chain classification based on two classified initial scenes did not significantly improve with scene 3 as a target scene compared to the across track chains 1-2-3 (overall accuracy loss 2.41%) and 5-4-3 (overall accuracy loss 1.62%) (Tables II-3 and II-5). However, when scene 4 was the target scene, two-sided chain classification did perform better compared to the corresponding tests 2-3-4 (overall accuracy loss 2.38%) and 6-5-4 (overall accuracy loss 6.07%). These results suggest that classification accuracy can be enhanced by using two overlap areas for the target scene when one-sided chain classification does not yield optimum results. This strategy may furthermore be considered when a target scene is located in a heterogeneous landscape and is not well described by the training data of only one overlap area.

As is the case for all land-cover classifications, training data should be well distributed and cover all characteristic landscape features across a scene. Due to the limited amount of Quickbird images in some regions of our study area and the lack of other ground truth data, reference data could not always be acquired with optimal distribution (Figure II-2). The higher variance in accuracy loss within the across-track tests are hence likely due to inhomogeneously distributed reference data.

Overall accuracy of the reference classifications was 96.26% on average. The stable performance across scenes with a limited amount of training data supported our assumption that SVMs are an appropriate classifier for chain classification. Nevertheless, chain classification is not restricted to SVM as a classifier and may be applied with other classifiers as well.

In chain classification, each scene is classified individually, with training data that is unique to this scene, and which is derived from the classified overlap area with its neighboring scene. This means that no signature extension from one scene to another is carried out and radiometric correction or normalization procedures are not required. This

greatly simplifies image processing and eliminates a potential error source. Examining illumination differences among scenes, chain classification in combination with SVMs was robust and no illumination influence was found. It is important to note though that successful chain classification in mountainous areas requires accurate orthorectification to account for differences in viewing geometry between neighboring scenes. And screening and masking of clouds and cloud shadows is necessary to avoid gross classification errors.

The good performance of chain classification in various tests suggests that it is a valuable approach for large area land-cover classifications with Landsat data. However, chain classification is by no means the only approach for this task, and it is important to understand advantages and disadvantages. In signature extension, radiometric calibration or normalization between images is an important preprocessing step (Pax-Lenney et al. 2001). This is not necessary when applying chain classification. The advantage of signature extension though, is the ability to extend classifications between single images over large distances (Olthof et al. 2005), i.e. neighboring scenes are not mandatory.

On the other hand, mapping large areas based on single scene classification presumes extensive reference data for each image as well as comprehensive resources for individual scene labeling (Cihlar 2000). Chain classification uses only a small part of the image data set to classify large areas. Therefore, an interpreter can focus on few well-classified initial images from where the rest is chain-classified. However, unlike large area classification based on a single scene or image mosaics, chain classification can only be applied to regions that share most spectral features.

The ‘applied radiometric normalization’ method developed by Cohen et al. (2001) is also based on the use of overlap areas between scenes. However, here statistical models translate the desired attributes of interest from the source image to the destination image. In comparison, chain classification based on SVMs, a faster and more straightforward process of land-cover mapping, requires no radiometric adjustments. Based on the available reference data, SVMs generate an initial classification that serves for training the neighboring scene in the overlap area.

In summary, chain classification is a promising new tool for large area land-cover classification. The approach is simple in that it only requires accurate georeferencing of scenes and no atmospheric correction. The accuracy loss of our classification was low (about 5% or less), even when long chains were classified. Chain classification is particularly well suited in areas where training data is only available for few scenes. This is

the case, for example, in areas of different forest ownership and hence different base map availability (e.g., state forest versus private forest), in many remote areas, or in places that are inaccessible or lack high resolution information for any other reason. Chain classification is much faster and lower in work load than single scene classification, but more limited in the total area that can be classified compared to signature extension and mosaic classification. Chain classification is not restricted to any sensor as long as enough overlap area between scenes is provided and the land-cover is homogenous across the images. An issue to be considered in future chain classification approaches is the use of a “hybrid” training sample collection to overcome the limitations set by the need of homogeneity. Here, in the case of missing or insufficient representation of classes in the overlap area, additional training samples would be collected manually in the target scene and added to the SVM chain classification procedure. Chain classification could also be used to classify images from different sensors, radiometric or geometric resolution, in the same chain as long as enough overlap area exists. In this study we applied chain classification to assess forest cover, but any other cover type can be considered as well, providing the potential for further applications in the context of large area land-cover mapping.

Acknowledgements

We gratefully acknowledge support by the Humboldt-University Berlin and by the NASA Land-Cover and Land-Use Change Program for this research. S. Schmidt and two anonymous reviewers provided valuable comments that greatly improved this manuscript.

Chapter III:
**Forest restitution and protected area
effectiveness in post-socialist Romania**

Biological Conservation (2012) in press

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Abstract

The effectiveness of protected areas can diminish during times of pronounced socio-economic and institutional change. Our goals were to assess the effectiveness of Romanian protected areas at stemming unsanctioned logging, and to assess post-socialist logging in their surrounding landscapes, during a time of massive socio-economic and institutional change. Our results suggest that forest cover remained fairly stable shortly before and after 1990, but forest disturbance rates increased sharply in two waves after 1995 and 2005. We found substantial disturbances inside protected areas, even within core reserve areas. Moreover, disturbances in the matrix surrounding protected areas were even lower than inside protected area boundaries. We suggest that these rates are largely the result of high logging rates, triggered by rapid ownership and institutional changes. These trends compromise the goals of Romania's protected area network, lead to an increasing loss of forest habitat, and more isolated and more fragmented protected areas. The effectiveness of Romania's protected area network in terms of its ability to safeguard biodiversity is therefore most likely decreasing.

1 Introduction

The world has failed to meet the 2010 target of halting biodiversity decline and species continue to be lost at an alarming rate (Butchart et al. 2010; CBD 2010; Hoffmann et al. 2010). Protected areas are the cornerstone of conservation efforts (Myers et al. 2000; Joppa et al. 2008; Cantu-Salazar and Gaston 2010), but land-use changes can trigger habitat loss and fragmentation outside protected areas, and may affect ecosystem processes within them (Hansen and DeFries 2007). Analyzing land-use changes in and around protected areas is therefore critical for assessing the effectiveness of this common biodiversity conservation strategy.

Protected area effectiveness is often compromised during periods of rapid socio-economic or institutional change, which can trigger widespread land-use changes and predatory resource use (Dudley et al. 2002; Irland 2008). For example, the collapse of socialism in the former Soviet bloc and transitions from planned to market economies generated drastic land-use changes (Ioffe et al. 2004; Kuemmerle et al. 2007; Baumann et al. 2011). At the same time, much of the infrastructure for nature protection eroded (Wells and Williams 1998), institutions weakened, and illegal logging and poaching increased (Soran et al. 2000; Vandergert and Newell 2003; Henry and Douhovnikoff 2008). More recently, a substantial number of Eastern European countries joined the European Union (EU), requiring them to significantly enlarge their protected area network (Oszlanyi et al. 2004; Young et al. 2007b). How these trends have affected the effectiveness of the regions' protected areas, however, remains poorly understood.

The Carpathians in Eastern Europe are of outstanding importance for nature conservation. The region has remained relatively undisturbed compared to Western Europe, is rich in biodiversity, and provides a refuge for large mammal populations (UNEP 2007; Anfodillo et al. 2008). It comprises Europe's largest mountain range and also largest continuous temperate forest ecosystem (UNEP 2007).

In some Carpathian countries, most notably Romania, large areas of forest land shifted from public to private ownership, including areas officially residing within protected areas. Implementing sustainable forest management and EU nature protection regulations in this new multi-ownership landscape is a formidable challenge (Strimbu et al. 2005). Yet, how logging rates and patterns have changed during the transition from socialism to market-

economies, and how forest ownership changes have affected protected area effectiveness in the Carpathians remains unexplored.

Assessing the status of forests in this region is often impaired by outdated forest resource information. In Romania, the last national forest inventory was carried out in 1984 (Brandlmaier and Hirschberger 2005; Marin et al. 2010). The lack of information about forest change is worrisome because Romania has some of Europe's last and most extensive old-growth, primary forests (400,000 ha in 1984; remaining 218,500 ha in 2004) (Veen et al. 2010) and harbors the largest European populations of brown bear (*Ursus arctos*), grey wolf (*Canis lupus*), and lynx (*Lynx lynx*) (Ioras et al. 2009).

Moreover, Romania's protected area network has undergone several fundamental changes following the collapse of socialism in 1989 (Soran et al. 2000; Oszlanyi et al. 2004; Ioja et al. 2010). Most importantly, Romania has implemented the EU Birds- and Habitat Directive (NATURA 2000), aimed at enlarging and connecting protected area networks. Today, about 20% of Romanian territory and about 10% of the country's forests are under some form of protection, including 13 national parks and 14 nature parks (Ioja et al. 2010). Most of Romania's protected areas are managed by the National Forest Administration Romsilva (Abrudan et al. 2009). While the recent increase in protected areas is a milestone for biodiversity conservation in Romania, considerable concerns about the status of nature protection remain: protected areas are sometimes subject to illegal logging and poaching, and many protected areas lack professional management, financing, and scientific support (Soran et al. 2000; Ioja et al. 2010).

Satellite images, particularly those from the Landsat Thematic Mapper (TM) and Enhanced Thematic Mapper Plus (ETM+) sensors, offer great opportunities to assess the effectiveness of protected areas, because they can capture forest disturbance (defined here as full canopy removal due to either natural disturbances, such as wind or insects, or anthropogenic disturbances, such as logging) and allow for robust comparisons across protected area boundaries (Young et al. 2006; Fraser et al. 2009; Huang et al. 2009). Since forest cover is correlated with species habitat and carbon storage (Keeton et al. 2010), forest disturbance is an indirect indicator for protected area effectiveness (Joppa and Pfaff 2010). We thus use Landsat images to conduct the first assessment of how forest changes have affected protected areas in Romania. We used satellite images to measure forest disturbance and compared disturbance rates inside and outside protected areas across a

30,000 km² region in northern Romania. Specifically, we ask the following research questions:

- What were the rates and spatial patterns of forest cover change in the post-socialist period (1989-2009)?
- Were protected areas effective in safeguarding the forests from logging within their boundaries?
- What was the effect of forest restitution on logging rates and patterns?

2 Study area

Our study area was one Landsat-scene footprint (34,225 km²) in the northern part of Romania, bordering Ukraine (Figure III-1). Elevations in the study region range from ~200 m to >2300 m above sea level (a.s.l.) and climate is transitional temperate-continental. Natural vegetation occurs in altitudinal belts (Donita and Roman 1976).

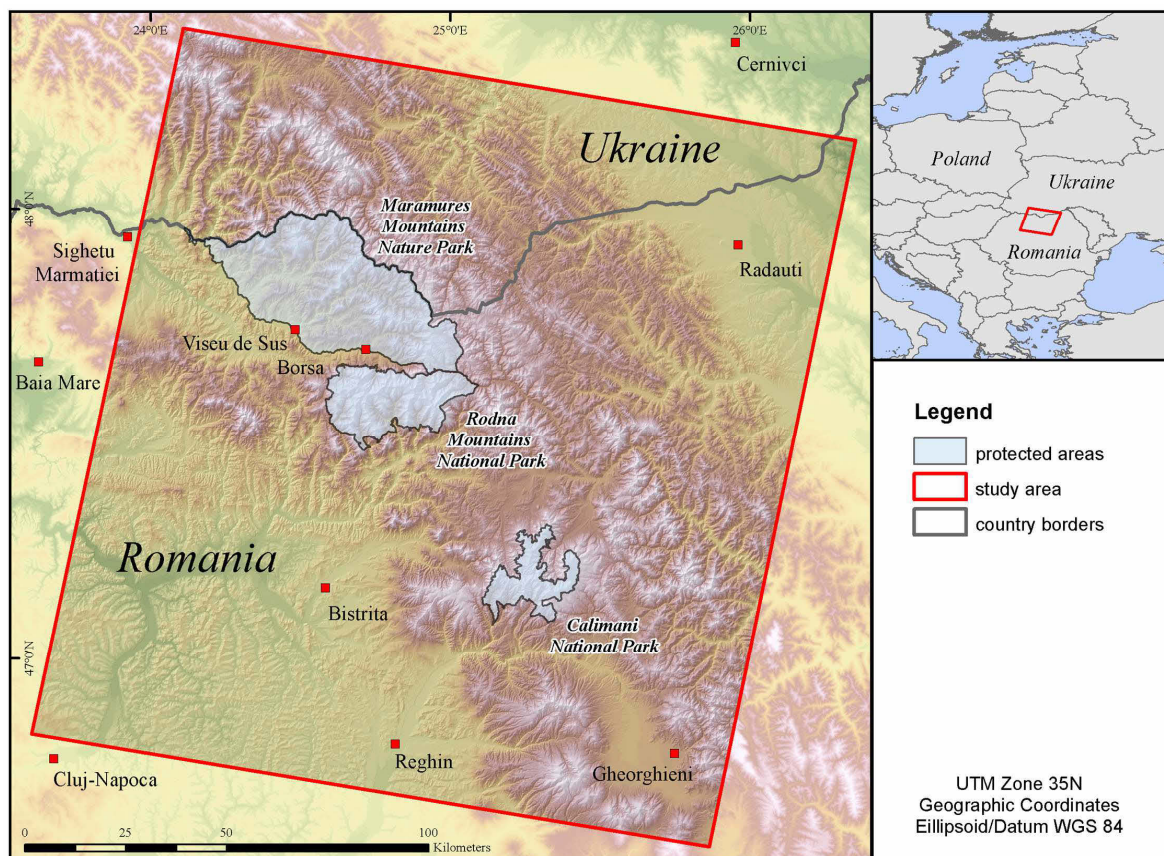


Figure III-1: Study area in the Carpathian Mountains in Eastern Europe including the three protected areas Maramures Mountains Nature Park, Rodna Mountains National Park, and Calimani National Park (Data: SRTM digital elevation model, ESRI Data and Maps Kit).

A deciduous forest belt consists of four sub-belts (from low to high altitudes): a Sessile oak (*Quercus petraea*) belt (300-600 m a.s.l.); a European beech (*Fagus sylvatica*) - durmast belt (600-1000 m a.s.l.); and a mixed forest belt with beech, Silver fir (*Abies alba*) and Norway spruce (*Picea abies*) (1000-1200 m a.s.l.). Higher altitudes (up to 1800 m a.s.l.) encompass a Norway spruce belt. Above the timber line, a sub-alpine belt (1800-2000 m a.s.l.) with dwarf pine (*Pinus mugo*) and juniper (*Juniperus communis ssp. nana*), and an alpine belt (>2000 m a.s.l.), dominated by dwarf shrubs and short-grass meadows prevails (Cristea 1993; Muica and Popova-Cucu 1993; Feurdean et al. 2007).

Our study region contains three large protected areas that collectively cover about 204,500 ha: Maramures Mountains Natural Park, Rodna Mountains National Park, and Calimani National Park. All three reserves are forest-dominated, but differ in their management history, size, and protection status. Maramures Mountains Natural Park (hereafter referred to as “Maramures”, ~150,000 ha) was established in 2004 and is Romania’s second largest protected area. It is embedded in the Maramures Mountains in Romania’s North, consists of 66% forests, 30% meadows and alpine pastures, and 4% agricultural land. As an IUCN category V protected area, limited human activities are allowed inside Maramures. About 19,000 ha are strictly protected in a so called ‘integral protection zone’. Rodna Mountains National Park (hereafter “Rodna”, 46,400 ha) was established in the 1930s, as a 183-ha protected area around the Pietrosu Mare peak. After several extensions, the park now covers almost the entire Rodna Mountain Range, and has been officially administrated since 2004 (Szabo et al. 2008). About 60% of Rodna is forested, while alpine pastures and dwarf pine cover most of the remaining area. Rodna falls into IUCN category II (Dumitras and Pop 2009) and has a core zone of about 20,800 ha (APNMR 2010). Calimani National Park (hereafter “Calimani”, 24,000 ha) was officially created in 1990 but did not become operative until 2003 (Toader and Dumitru 2005). It is part of the Calimani Mountains, the least populated montane region in Romania. Calimani is also classified as IUCN category II and is comprised of 58% old-growth Norway spruce and mixed beech-conifer forest. The core zone (i.e., strictly protected areas) covers an area of about 16,800 ha.

Romania has restituted (i.e. privatized) almost 45% of its forests prior to 2009 over the course of three phases based on laws passed in 1991, 2000, and 2005 (Abrudan et al. 2009; Strimbu et al. 2005). At the end of the restitution process, about two-thirds of all forest will be in private ownership, corresponding to about 800,000 new forest owners (Ioras and Abrudan 2006). Since forests were restituted during a period of economic hardship and

weak political institutions, incentives for new owners to capitalize on their forest land by over harvesting were high (Turnock 2002; Nichiforel and Schanz 2011).

3 Materials and methods

3.1 Datasets used

We used eight mid-summer to early-autumn Landsat TM/ETM+ images from path/row 185/27 to reconstruct forest disturbance histories from 1987 to 2009. We acquired images mainly before and after restitution laws were passed, and before and after protected areas were established (images from 8 Sept. 1994, 4 July 2002, 4 Sept. 2004, 11 Oct. 2006, and 15 July 2009) and complemented our time series with images spanning the time period of 1986 to 1989 (18 Sept. 1986, 7 Oct. 1987, and 8 July 1989) to establish a baseline representing the forest cover of the late socialist period. Landsat images have 30-m resolution and are well-suited for mapping forest cover changes in the Carpathians at landscape scales (Kozak et al. 2008; Main-Knorn et al. 2009; Olofsson et al. 2011). Of the seven spectral bands, the thermal band was not retained for our analysis. Seven images had already been orthorectified by the United States Geological Survey. One image was obtained from the Global Land Cover Facility (www.landcover.org) and co-registered to the other images. We masked clouds, and cloud shadows (1986: 2572 km²; 1987: 2029 km²; 1989: 1062 km²; 1994: 983 km²; 2002: 1607 km²; 2004 no clouds; 2006: 4897 km²; 2009: 338 km²).

Reference data for training and validation was collected based on high resolution satellite images or air photos available in Google Earth that cover the complete study area (Knorn et al. 2009; Baudron et al. 2011). We sampled 3000 random points and classified those as either forest or non-forest based on visual interpretation. Points were considered forested if tree cover exceeded 60% and forest patches were larger than one Landsat pixel (900 m²) (Kuemmerle et al. 2009). Our forest definition thus included orchards (almost absent from our study region), hedgerows, or open shrubland, but not areas with isolated trees. All points were cross-checked visually with the Landsat images to ensure that class labels did not change between 1986 and 2009. Points located in areas covered by clouds or with unclear class membership were discarded. In total, we used 2604 reference points (1280 non-forest, and 1324 forest).

Additional spatial data included an enhanced digital elevation model based on the Shuttle Radar Topography Mission (SRTM, <http://srtm.csi.cgiar.org>), resampled from 90 to 30 m to match the spatial resolution of the Landsat images. We also obtained protected area boundaries (provided by the National Forest Administration of Romania, Protected Areas Department), administrative boundaries (ESRI Data and Maps Kit 2008), core protected zones (provided by the protected area administrations), and areas of old-growth forest (provided by the Romanian Forest Research and Management Institute - ICAS). Extensive field visits and interviews with protected areas staff and park administrations, stakeholders, NGOs as well as several researchers were carried out in 2008, 2009, and 2010. Field visits served to photo-document and geo-locate examples of forest disturbance sites to identify the causes of these disturbances, facilitated also by local knowledge provided by project partners.

3.2 Forest disturbance mapping

We used the forest disturbance index (Healey et al. 2005; Kuemmerle et al. 2007) to map forest cover changes in our study area. Our analysis consisted of two steps. First, we classified forest and non-forest areas for the late 1980s (three images) and 2009 (one image) using Support Vector Machines (SVM) (Knorn et al. 2009; Kuemmerle et al. 2009). It was necessary to use three images for the 1980s to obtain an area-wide map due to high cloud coverage in each of these images. SVM delineates two classes by fitting a separating hyperplane based on training samples. This hyperplane is constructed by maximizing the margin between class boundaries and is described by a subset of training samples, so-called 'support vectors' (Boser et al. 1992; Cortes and Vapnik 1995; Foody et al. 2007). SVM require training data that optimize class separation rather than describing the classes themselves (Foody and Mathur 2006). Using a radial basis function, class distributions with non-linear spectral feature space boundaries can be mapped into a higher dimensional space for linear separation (Huang et al. 2002). A mathematical description of SVM can be found in Huang et al. (2002).

To train and validate the SVM classifier, we used ten-fold cross-validation, where we split all available reference points into training (90%) and validation (10%) samples. We classified each of the four images (1986, 1987, 1989, and 2009) for all possible splits (i.e., 10 times), calculated accuracy measures for each run, and averaged the error estimates (Steele 2005; Knorn et al. 2009). We calculated overall accuracy, kappa value, and class-wise user's (error of commission - a pixel is assigned to an incorrect class) and producer's

(error of omission - a pixel is omitted from its correct class) accuracies (Congalton 1991; Foody 2002). The final forest/non forest classifications were based on all reference points (Burman 1989).

We used the forest/non-forest classifications to generate a forest land map by masking all permanent non-forest areas (i.e., non-forest in the late 1980s and 2009). Forests disturbed immediately prior to the acquisition of our earliest image (1986) and which had regenerated by 2009 were thus not assigned to the permanent non-forest class. This means, that areas that were disturbed before 1986 but forested in 2009 are defined as forest land while appropriately assigning the respective disturbances to the late socialist time period. For the resulting forest land map, we used a minimum mapping unit of ~1 ha (10 pixels) based on high-resolution satellite image interpretation and extensive field visits.

Second, we calculated the disturbance index for all forest land pixels and for each image in our time series. The disturbance index is a continuous index based on the Tasseled Cap transformation and emphasizes the difference in spectral signatures between stand-replacing disturbance (high disturbance index values) and all other forest features (low disturbance index values). The DI uses the Tasseled Cap indices by making use of spectral differences between undisturbed forest (high greenness and wetness components, low brightness) and recently disturbed forests (low greenness and wetness, high brightness). Calculating the DI, requires two types of information: first, a forest and non-forest map, and second, the normalization of each Tasseled Cap component relative to the typical reflectance properties of undisturbed forests. Using the three normalized components the DI is calculated as the brightness minus the sum of greenness and wetness. Separating disturbed from undisturbed forests requires setting a disturbance index threshold for each image. To define this threshold, we randomly selected 30 locations and digitized on screen the two closest disturbances as polygons in each of the Landsat images. Thresholds were determined by extracting the disturbance index range describing the digitized disturbances and setting a disturbance index threshold at the lowest quartile of this distribution. This rather conservative approach was chosen to avoid errors of commission (i.e., overestimation). The result yielded a forest disturbance map for 1987-2009 with the disturbance classes '1987-1989', '1989-1994', '1994-2002', '2002-2006' and '2006-2009'. For this map we used a minimum mapping unit of ~0.4 ha (i.e., 4 pixels) and we excluded disturbances above 1600 m that mainly represented misclassifications due to phenology effects. We then assessed the total disturbed area (in ha) and the annual disturbance rate (in %) for each time period. To validate our final forest disturbance map, we used a

stratified random sample of 50 points per disturbance class and 150 points for the permanent forest and permanent non-forest class, respectively. We complied a minimum distance of 1000 m between points to limit spatial auto-correlation. All points were photo-interpreted using Google Earth and the Landsat images (Knorn et al. 2009; Kuemmerle et al. 2009). Finally, an error matrix including area-adjusted user's and producer's accuracies as well as overall accuracies were calculated considering the true area proportions of each class (Card 1982). Additionally, we calculated 95% confidence intervals around our area estimates (Cochran 1977).

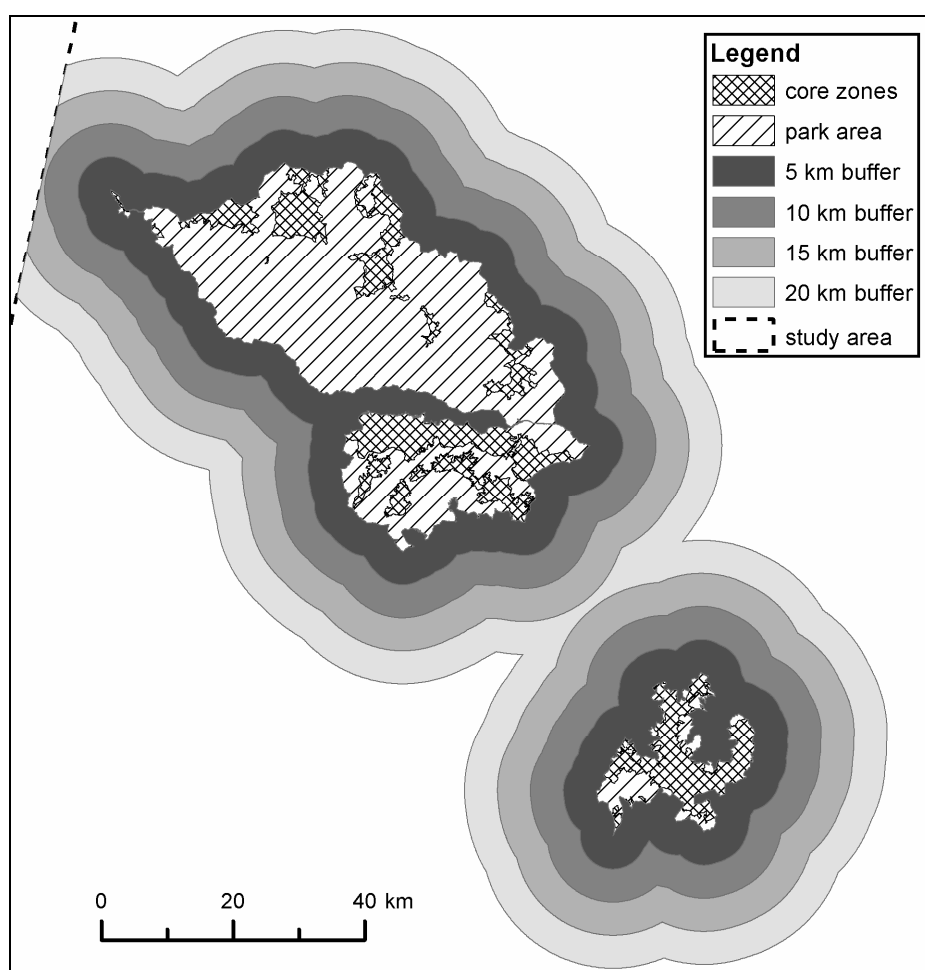


Figure III-2: Buffers and zones of protected areas used to summarize forest disturbance rates.

To assess the effectiveness of protected areas, we summarized disturbances inside and outside the protected areas by calculating annual disturbance rates separately for each zone. Inside protected areas, we also distinguished between core (strictly protected) and non-core areas. Outside protected areas we assessed disturbance rates in 5 km buffer zones within 5, 10, 15 and 20 km distance, respectively. We delineated the buffer zones for all protected areas together, i.e. buffers intersecting between neighboring protected areas were merged, thus ensuring that each disturbance was assigned only once to a single buffer zone

or protected area (Figure III-2). To assure comparability of disturbance rates between the protected areas and the surrounding buffers, widths of the buffer zones were determined according to the amount of forest land found in all three protected areas summarized (Table III-2)

4 Results

The SVM classification resulted in reliable forest/non-forest maps for the individual years, with overall accuracies generally exceeding 90% (1986: 93.4%; 1987: 93.2%; 1989: 92.3% and 2009: 94.6%) and kappa values exceeding 0.85 (1986: 0.87; 1987: 0.86; 1989: 0.85 and 2009: 0.89). The change detection based on the forest disturbance index also yielded a reliable forest disturbance map, with an overall accuracy of 94.9% and relative narrow confidence intervals around the area estimates (Table III-1).

Table III-1: Error matrix for the forest disturbance map including area-adjusted user's / producer's accuracies together with mapped and adjusted areas and the 95% confidence intervals.

	Prod. acc	User's acc	Map area (ha)	Adj area (ha)	±95% CI (ha)	±95% CI (%)
Dist. 1987	94,63%	94,00%	30,216	30,015	3049	10,16%
Dist. 1989	96,84%	92,00%	40,309	38,293	3951	10,32%
Dist. 1994	100,00%	96,00%	60,441	58,023	3384	5,83%
Dist. 2002	99,47%	90,00%	375,802	340,036	32,277	9,49%
Dist. 2004	98,48%	90,00%	30,241	27,638	2726	9,86%
Dist. 2006	100,00%	92,00%	21,060	19,375	1632	8,43%
Dist. 2009	100,00%	94,00%	188,849	177,518	12,814	7,22%
Forest	94,77%	95,33%	14,331,822	14,417,346	708,756	4,92%
Non-forest	94,87%	94,67%	13,756,898	13,727,393	708,949	5,16%

Forests covered about 59% of the study region and forest disturbances were widespread between 1987 and 2009, especially for the period 1994 to 2002 (about 1.7% of the forest land; 30,742 ha) and 2006 to 2009 (about 0.95% of the forest land; 16,993 ha). In total, 60,945 ha of forest were disturbed over the 22 year time period we studied. Annual disturbance rates were highest between 2006 and 2009 (0.32%; 5664 ha/year).

We found substantial forest disturbance both inside and outside the three protected areas during all time periods (Table III-2). In total, 7288 ha of forest cover were lost between 1987 and 2009 in the three protected areas (4.6% of the forest land). This is higher than the disturbances found in the respective buffer zones (Figure III-2) (5 km: 4.0% [6107 ha]; 5-10 km: 3.3% [4617 ha]; 10-15 km: 3.5% [5277 ha] and 15-20 km: 3.3% [5270 ha]). The amount of disturbance differed markedly between time periods though. For instance,

disturbance rates for all protected areas were relatively low between 1987 and 1994 (<0.10%), but increased almost 10-fold between 1994 and 2002. This pattern repeated after 2002, with low disturbance rates between 2002 and 2006 (<0.12%) followed by an 8-fold increase after 2006. Of the total disturbed area, 4229 ha (2.69% of the forest land) were disturbed between 1994 and 2002 and 2075 ha (1.32% of the forest land) were disturbed between 2006 and 2009 (Table III-2). However, parts of these disturbances occurred before the official recognition of the protected areas (Maramures in 2004, Rodna and Calimani in 2003). With more than 4800 ha disturbed in 22 years, Maramures had the largest amount of total disturbed area. Moreover, with a disturbance rate of 0.56% between 2006 and 2009 this is the highest of all parks across all time periods. In Rodna, highest disturbances rates occurred between 1994 and 2002 (0.47%), when they were 3-times above its annual average and the highest of the three protected areas during this period. Similar to Rodna, the highest disturbance rates for Calimani were found between 1994 and 2002 (0.43%), and the second highest between 2006 and 2009 (0.20%) (Table III-2).

Table III-2: Disturbances per protected area zone, buffer zone and time period. Numbers correspond to disturbed area (in ha) and yearly rates in relation to forest land (in %). MMNP = Maramures Mountains Nature Park; RMNP = Rodna Mountains National Park; CNP = Calimani National Park.

	1987 - 1989	1989 - 1994	1994 - 2002	2002 - 2004	2004 - 2006	2006 - 2009	sum	forest land (ha)	
MMNP	<i>Core zone</i>	16 (0.06%)	21 (0.03%)	198 (0.19%)	4 (0.02%)	26 (0.10%)	87 (0.23%)	352	12,736
	<i>Non-core zone</i>	169 (0.09%)	283 (0.06%)	2057 (0.28%)	133 (0.07%)	209 (0.12%)	1629 (0.60%)	4481	90,253
	<i>Entire park</i>	185 (0.09%)	304 (0.06%)	2254 (0.27%)	138 (0.07%)	235 (0.11%)	1716 (0.56%)	4833	102,989
RMNP	<i>Core zone</i>	14 (0.04%)	14 (0.02%)	782 (0.61%)	3 (0.01%)	4 (0.01%)	47 (0.10%)	864	16,093
	<i>Non-core zone</i>	13 (0.04%)	24 (0.03%)	463 (0.34%)	13 (0.04%)	7 (0.02%)	187 (0.36%)	707	17,204
	<i>Entire park</i>	27 (0.04%)	38 (0.02%)	1245 (0.47%)	16 (0.02%)	10 (0.02%)	235 (0.24%)	1571	33,296
CNP	<i>Core zone</i>	10 (0.04%)	1 (0.00%)	230 (0.21%)	1 (0.00%)	0 (0.00%)	38 (0.09%)	280	13,718
	<i>Non-core zone</i>	5 (0.04%)	7 (0.02%)	499 (0.85%)	4 (0.03%)	2 (0.00%)	86 (0.39%)	604	7324
	<i>Entire park</i>	16 (0.04%)	8 (0.01%)	729 (0.43%)	5 (0.01%)	2 (0.00%)	124 (0.20%)	884	21,042
Sum all parks	227	351	4229	158	247	2075	7288	157,327	
Buffer zone	<i>5-km buffer</i>	382 (0.09%)	583 (0.03%)	2473 (0.28%)	102 (0.04%)	121 (0.04%)	2446 (0.80%)	6107	153,040
	<i>10-km buffer</i>	450 (0.16%)	544 (0.04%)	1950 (0.29%)	130 (0.06%)	184 (0.09%)	1358 (0.46%)	4617	138,330
	<i>15-km buffer</i>	410 (0.09%)	566 (0.04%)	2558 (0.36%)	187 (0.07%)	155 (0.05%)	1402 (0.28%)	5277	151,726
	<i>20-km buffer</i>	306 (0.07%)	458 (0.04%)	2590 (0.26%)	202 (0.05%)	174 (0.03%)	1541 (0.38%)	5270	158,848

In total, 4.20% of the forest land in Calimani (884 ha) was disturbed between 1987 and 2009, compared to 4.72% (1571 ha) and 4.69% (4833 ha) in Rodna and Maramures, respectively (Table III-2). For the time period following the establishment of the three

protected areas (2003/2004–2009), disturbance rates were generally lower in the core zones compared to the rest of the park (0.17% [113 ha] for Maramures, 0.06% [51 ha] for Rodna, 0.06% [38 ha] for Calimani).

5 Discussion

Rapid socio-economic changes due to the transition from socialism towards a market-economy triggered forest disturbances and illegal resource use even inside protected areas. Our results help to address the question of how forest cover in the Romanian Carpathians has changed after the collapse of socialism, and how this, in turn, may have affected the ability of Romania's protected areas to safeguard biodiversity. Our remote sensing analysis indicated widespread forest cover changes between 1987 and 2009, especially since 2006. Disturbances inside protected area boundaries were even higher than those in their surrounding. While our remote sensing based approach cannot distinguish between natural and anthropogenic disturbances, our results, field visits and interviews suggest that natural disturbances alone do not explain the increasing trend in forest loss. We suggest that the ongoing forest restitution process and associated harvesting were a major underlying cause for the accelerated disturbance rates observed (Griffiths et al. 2012).

Massive socio-economic transformations accompanied by substantial economic hardship, and the restitution process translating into logging thus present considerable challenges for nature conservation. The observed disturbance rates show that the effectiveness of the three protected areas is challenged, and forest disturbance is both compromising habitat integrity within protected areas and may be fragmenting their surrounding landscapes. Since forest loss close to protected areas can affect ecosystem functions and processes, hamper species dispersal, or induce edge effects (Cameron 2006; DeFries et al. 2010), protected area management and conservation planning should consider that parks are embedded in larger landscapes which are important for conservation. While Romania now has an extensive network of parks that appear "protected on paper", continued monitoring of these parks is necessary to ensure their effectiveness. As shown in our analysis, satellite image interpretation can contribute substantially to this task.

Natural stand-replacing disturbance events occur in the Romanian Carpathians and include insect infestation, avalanches and wind-throw, with the latter being the most important (Schelhaas et al. 2003; Toader and Dumitru 2005). Forest fires are not widespread and cause negligible disturbances (0.15% of the Romanian forest area in 1965-1998) and are

always confined to small patches (Anfodillo et al. 2008; Rozyłowicz et al. 2011). Nevertheless, natural disturbances are unlikely to explain the forest cover change trends we observed. Whereas some large-scale natural disturbances occurred in our study region, wind disturbances often affect regions much smaller than our minimum mapping unit (Rozyłowicz et al. 2011). Moreover, natural disturbances cannot explain the strong increase in forest disturbances we observed after 2006. Indeed, wind-throw events occur across the Carpathians, but with varying frequency (Lavnyy and Lässig 2007) and for Romania with a declining frequency and intensity since 1975 (Popa 2008).

Wind-throw events or insect outbreaks are most frequent in artificial spruce plantations (Keeton and Crow 2009; Kuemmerle et al. 2009; Macovei 2009), that often comprise non-native genetic spruce variations, and thus are related to forest management history (Schelhaas et al. 2003). Moreover, intensive exploitation in the past simplified forest structure and composition at stand and landscape scales, resulting in fragmentation and high contrast forest edges that increase vulnerability to wind-throw (Toader and Dumitru 2005; Macovei 2009). Many forest cover changes classified as natural disturbances may therefore actually be anthropogenic in origin. Likewise, this evidence suggests that wind-throw events should be at least equal in areas outside of reserves which have more substantial forest management histories.

Corruption and lack of transparency is also a major problem, leading to cases where sanitary or salvage logging has been misused to harvest healthy forest stands (Brandlmaier and Hirschberger 2005). Informal, interviewees have even pointed out to us during field work that corridors in forests were deliberately placed to inflict wind-throw and thereafter allow for salvage logging. In sum, although we cannot separate natural disturbances and logging based on satellite data alone, true natural disturbances are rare in the Carpathians and natural disturbances neither explain the increase in disturbance rates since 1989, nor the differences in disturbance rates inside and outside protected areas.

Instead, we suggest that the major institutional and socio-economic changes relate to the high rates of disturbance during post-socialism compared to disturbance rates observed during the last years of socialism. We caution that a causal connection cannot be established, as spatially-explicit ownership data on forest ownership is currently not available. However, our results, extensive field-visits, expert interviews and other studies from other areas in Romania (Griffiths et al. 2012) all unanimously suggest that the

disturbance trends we observed are indeed due to the changes in forest legislation (Irimie and Essmann 2009; Mantescu and Vasile 2009).

New owners appear to harvest much of their forests to gain short-term profits. Moreover, new forest owners often lack of capacity and knowledge for sustainable forest management and nature conservation principals and legislation. New forest owners additionally often doubt the permanence of their newly gained property rights and there is a lack of knowledge on sustainable harvesting principles (UNDP 2004). Additionally, cases of illegal logging in restituted forests brought to court often remain unpunished or are left with inadequate consequences (pers. comm., local scientists)*². In consequence, widespread logging and over-harvesting was evidenced after the first restitution law in 1991 (Nichiforel and Schanz 2011). Most of the restituted forest were immediately cleared by new owners (Mantescu and Vasile 2009). Similar trends occurred in the subsequent restitution phases following the respective laws in 2000 and 2005 (Ioras et al. 2009), amplified by weakened institutions and increasing economic hardship. The effectiveness of the three protected areas we studied is in question. Since Maramures is one of the poorest regions of Romania and more than 24,000 ha (16% of the park area) has been restituted, habitat fragmentation and degradation due to clear-cutting and unsustainable forest management is a major threat (UNDP 2004). Accordingly, our results show that frequent disturbances throughout Maramures, including old-growth forest (e.g., Figure III-3b, circle 1), took place since the collapse of socialism, even partly exceeding those outside the protected areas (Table III-2; Figure III-3). After 1989, the entire Maramures Mountains became a target of timber companies and timber harvesting is now the mainstay of the local economy (Munteanu et al. 2008).

One prominent example of logging exceeding the maximum allowed patch size of 3 ha is found in the upper Tibau Basin (Figure III-3a, circle 1; Figure III-3, photograph I), where a forest area of up to 500 ha was cleared between 2006 and 2009. This substantially increases flood vulnerability in the area, taking into account that extensive logging both outside and inside Maramures already contributed to severe flood events in the past (UNDP 2004; Munteanu et al. 2008). The lower protection status of the Maramures Nature

² Full names are not provided to protect interviewees and informants.

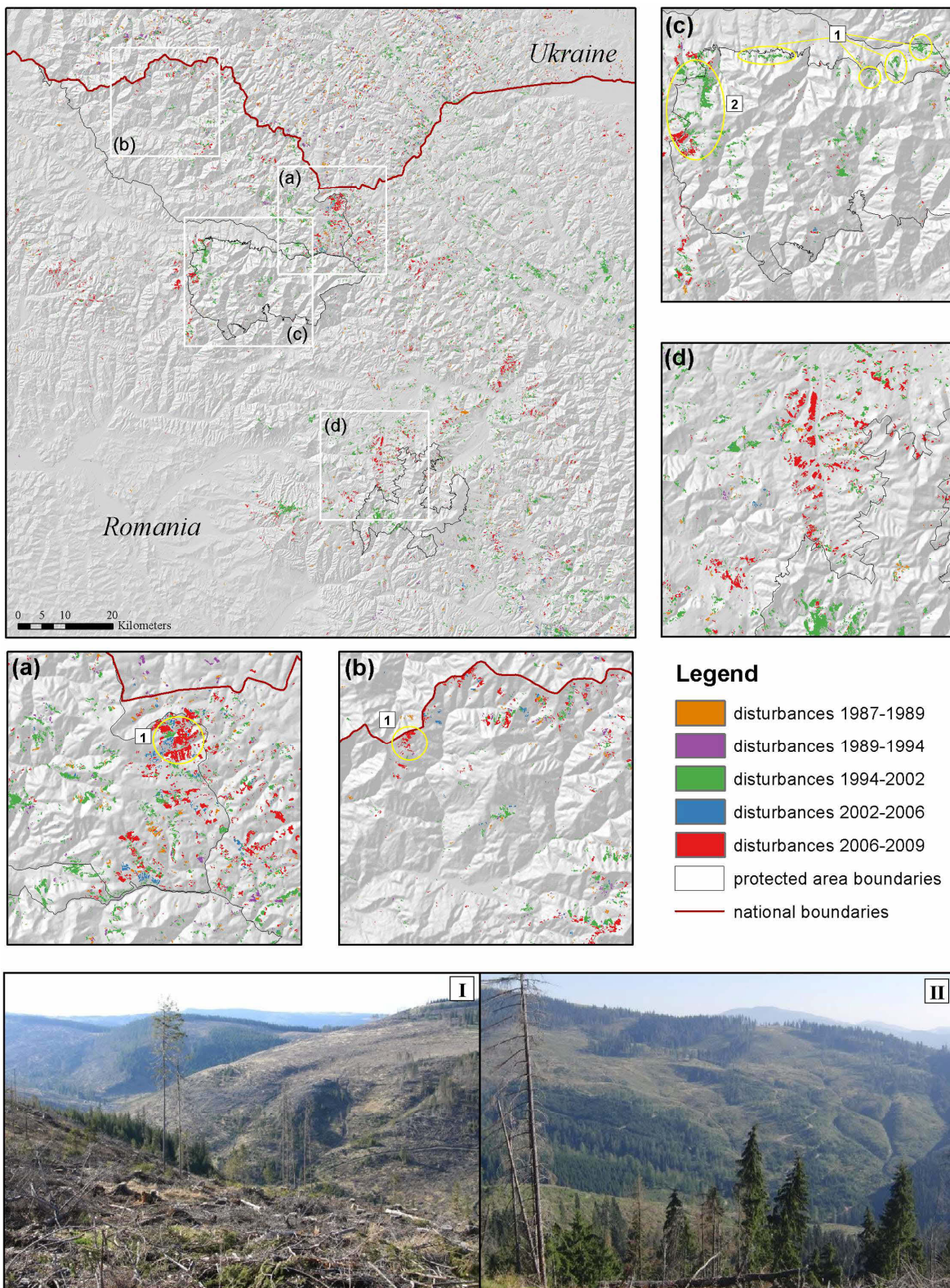


Figure III-3: Forest disturbances rates within and around the protected areas. Circles highlight disturbance events described in the text. (a) eastern part of Maramures, (b) northern part of Maramures, (c) western part of Rodna, and (d) western part of Calimani. Photographs of (I) forest loss in the Tibau basin in Maramures Mountains Nature Park, and (II) a wind-throw area in the western part of Rodna Mountains National Park (Photos: M. Mindtrescu; J. Knorn).

Park may further explain the highest disturbance rates of all three protected areas inside and outside the core zone (Table III-2). Taking into account that each of the rangers is responsible for patrolling on average almost 12,400 ha (Ioja et al. 2010) (while a forester is usually in charge of only 1000 ha), enforcing legislation remains challenging.

Forests inside Rodna are now owned by more than 20 entities. This fragmentation of ownership and management creates an extremely difficult situation for the park administration. It is encouraging, though, that the main proportion of forest disturbances in Rodna occurred before 2002 (Figure III-3). According to the park administration, parts of these disturbances are due to illegal logging (Figure III-3c, circles 1). This was the case, for example, in the Pietrosu Mare scientific reserve between 1995 and 2004. Due to an increased exposure, the remaining forest suffered additionally from wind-throw impacts and bark-beetle infestation (pers. comm. with park administration). Beside these logging events, our results clearly depict impacts of wind-throws in the western part of Rodna (Figure III-3c, circle 2; Figure III-3, photograph II).

Rodna presents a particularly striking example for the lack of appropriate buffer zones. The two scientific reserves Pietrosu Mare and Piatra Rea do not have a buffer area on the northern side of the park. Reasons for this originate in the history of the establishment of parks in Romania. Due to economic pressure and without knowledge of modern conservation planning principles, it was generally agreed that small protected areas are best for biodiversity conservation (Soran et al. 2000). One reason for high disturbance rates in the surroundings of the parks may thus originate from the absence of suitable buffer areas.

In Calimani, we found the least amount of forest disturbances and rates of all protected areas (Table III-2). However, recent forest disturbances increased substantially around the Calimani, likely contributing to an increased isolation of the park. Parts of the scattered disturbance patches in the western part of the protected area are the result of wind-throws between 1994 and 2002 (pers. comm. with park administration) (Figure III-3d).

Our study showed widespread forest cover changes in Romania since the breakdown of socialism, mainly due to excessive logging triggered by the recent forest restitution. Forest disturbances were even widespread within protected areas and old-growth forests, sometimes exceeding disturbance rates in the surrounding landscape. The root causes of increasing logging rates in the post-socialist period are economic hardships and a generally low awareness of the role of natural resources and biodiversity, particularly concerning non-market ecosystem services (e.g., flood protection) (UNDP 2004; Young et al. 2007b).

In addition, institutional decay, corruption, and an under-funded nature protection program further hamper the implementation of nature conservation legislation. The high amount of forest disturbances we found thus adds to recent voices of concern regarding nature protection in Romania (UNEP 2007). The Carpathians, and especially Romania, harbor unique high-conservation value forests that redevelop only very slowly (Ioras et al. 2009; Wirth et al. 2009a). Halting the ongoing loss of these forests requires capacity building and reinforcing Romania's nature protection infrastructure. In the short run, continued monitoring of forest losses and protected area effectiveness are needed, and satellite image analyses offers valuable tools for doing so.

Acknowledgements

We gratefully acknowledge support by the Alexander von Humboldt Foundation, the European Union (Integrated Project VOLANTE FP7-ENV-2010-265104) and the NASA Land-Use and Land-Cover Change Program of the National Aeronautics and Space Administration (grant number NNX09AK88G). A. Sieber, E. Stanciu, M. Turticà, V.N. Nicolescu, Dr. I. Blada, and the park administrations provided valuable input that greatly improved this manuscript. We thank A. Janz and S. van der Linden for implementing the imageSVM software (www.hu-geomatics.de). We also thank V. Butsic, three anonymous reviewers and the editor for constructive criticism and helpful comments.

Chapter IV:
**Continued loss of temperate old-growth forests
in the Romanian Carpathians despite an
increasing protected area network**

Environmental Conservation (submitted)

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Abstract

Old-growth forests around the world are vanishing rapidly and are almost completely lost in the European temperate forest region. Poor management practices, often triggered by socio-economic and institutional change, are the main causes for their loss. Some of the last remaining tracts of these forests within Europe are located in Romania. We used satellite image analysis to assess recent trends in old-growth forest cover there and our results suggest that their coverage declined by 1.3% from 2000 to 2010. Romania's protected area network has been expanded substantially with the country's accession to the European Union, and most of the remaining old-growth forests were located within them. Surprisingly though, 72% of the old-growth forest clearings were found within protected areas, highlighting the threats still facing these old-growth forests. It appears that logging in protected old-growth forests is at least in part related to institutional reforms, insufficient protection, and ownership changes. Thus, obviously causing a decrease in old-growth forest surface is the fact, that harvesting activities in most of these forests is accordance with the law. Without policy interventions at all institutional levels, the future of Romania's old-growth forests and the important ecosystem services they provide will remain uncertain.

1 Introduction

Across the globe biodiversity is declining and the “2010 target” of the Convention on Biological Conservation has not been met (Butchart et al. 2010; CBD 2010). The destruction and fragmentation of habitat along with overexploitation are the main causes of the global biodiversity crisis (MA 2005a; Brook et al. 2008; Ehrlich and Pringle 2008). Old-growth forests play a key role in maintaining biodiversity and are irreplaceable for sustaining biodiversity (Gibson et al. 2011). Moreover, contrary to the long standing view that old-growth forest are carbon neutral, they continue to sequester carbon for long time periods, but also store more carbon per unit area than any other ecosystem or forest successional stage (Luyssaert et al. 2008; Knohl et al. 2009; Wirth 2009; Keeton et al. 2011). Old-growth forests in the Carpathian Mountain region of Europe, in particular, store very high levels of carbon in comparison to younger and managed forests (Holeksa et al. 2009; Keeton et al. 2010).

Despite their ecological importance, old-growth forests around the world are vanishing at an alarming rate (Achard et al. 2009), thereby diminishing the ecosystem services (e.g., genetic resources, protection from natural hazards, riparian functionality) that they provide (Keeton et al. 2007; Wirth et al. 2009a) and threatening the biodiversity they harbour. Especially in the industrialised countries of northern Europe, land-use changes and conversion of primary forests to managed plantations have almost completely eradicated old-growth forests (Wirth et al. 2009b). Of the total forest area in central Europe, only 0.2% of old-growth forests survived mainly in remote, mountainous areas or within nature reserves (Frank et al. 2009; Schulze et al. 2009). Furthermore, human disturbances within these remaining old-growth forests continue and in many cases has had long-lasting negative effects on species composition and key habitat functions (Frank et al. 2009).

Goods and services from European temperate forests, such as clean water, wood products and recreation opportunities in relation to the large number of people living in close proximity, make these forests socio-economically important (Thompson et al. 2009). One area where forests are particularly valuable in this respect are the Romanian Carpathians, comprising the eastern and southern extension of the mountain range and approximately 50% of their total area. Carpathian old-growth temperate forests have high biodiversity and nature conservation value due to their high diversity in plant and animal species (UNEP

2007; Keeton et al. 2010). They are also an important foundation for local livelihoods, and for ecosystem goods and services in the region.

Assessing the status of old-growth forests in the Carpathians is difficult due to often outdated, incomplete and fragmented forest resource information. The last national forest inventory for Romania was carried out in 1984 (Brandlmaier and Hirschberger 2005; Marin et al. 2010). This lack of information is worrisome because Romania harbors some of Europe's last and still extensive old-growth tracts. A comprehensive assessment of the status of old-growth forests was performed there between 2001 and 2004 (Veen et al. 2010), identifying about 210,000 ha of old-growth forest and comprising about 3.5% of total Romanian forest cover. This is more than in any other Central European country. However, the extent of Romania's old-growth forest has decreased by 70% since 1945 and numerous severe threats have been identified, including illegal logging, invasive species, and climate change (Biriş and Veen 2005; Price et al. 2011).

While using the term "old-growth forests", we follow the suggestions and definitions proposed by Wirth et al. (2009b) and Burrascano (2010). These include widely accepted criteria for moist temperate old-growth forests: relatively old stand age, abundance of large old tree species, deadwood components (both standing and downed), dominance by late-successional tree species, vertically complex canopies, and horizontal structural heterogeneity (i.e. gap mosaics). These elements of stand structural complexity correlate with a variety of habitat functions for late-successional forests; these are frequently missing or under-represented in younger or managed forests (Keeton 2006; Smith et al. 2008).

In Romania, vast forests including these old-growth forest patches, provide important habitat for the largest European populations of brown bear (*Ursus arctos*), grey wolf (*Canis lupus*), and lynx (*Lynx lynx*). Moreover, these old-growth forests have been recognized for their exceptional biodiversity harboring many endemic, rare, and threatened forest species (Biriş and Veen 2005; Ioras and Abrudan 2006; Biriş et al. 2010). However, one of the most pressing recent threats relates to the changes in forest ownership pattern (Nijnik et al. 2009; Griffiths et al. 2012; Knorn et al. in press). Large areas of state forest have been restituted to prior owners, and often this has resulted in forest management changes (MCPFE 2007; Barbier et al. 2010; Lambin and Meyfroidt 2010). Economic hardship accompanied by weak political institutions encouraged land owners receiving restituted forests to liquidate their timber assets through harvesting (Turnock 2002;

Nichiforel and Schanz 2011). The combination of an uncertain institutional environment (Lambin et al. 2001), poverty, and the high timber value of old-growth forests additionally increases the pressure to exploit beyond sustainable dimensions (Anfodillo et al. 2008). Moreover, the fast growing number of small-scale forest holdings (around 800,000 by the end of the restitution process) (Ioras and Abrudan 2006), hampered the establishment of sustainable forest management practices and effective safeguarding of old-growth forests (Turner et al. 1996; Nijnik et al. 2009; Źmihorski et al. 2010). Last but not least, a weak law enforcement fosters logging practises and magnitudes outside legal norms (Brandlmaier and Hirschberger 2005; Knorn et al. in press). These continuing threats and losses necessitate a new assessment to update and validate the last inventory and to provide a basis for the effective conservation of these forests. Moreover, this assessment occurred before the major institutional reforms in the forest sector were completely implemented and the identification methods used in this inventory did not include a reporting or spatial description of pre-existing disturbance impacts.

Satellite image interpretation is the most accurate and comprehensive approach for assessing forest cover changes across large areas (Achard et al. 2009; FAO 2011). Especially images from the Landsat Thematic Mapper (TM) and Enhanced Thematic Mapper Plus (ETM+) sensors are able to capture canopy removal reliably across large regions (Young et al. 2006; Fraser et al. 2009; Huang et al. 2009), including for parts of the Carpathians (Kozak et al. 2008; Kuemmerle et al. 2009; Main-Knorn et al. 2009; Olofsson et al. 2011). Satellite analyses are particularly well suited to map forest disturbances, because the reflectance of a given pixel changes drastically when the structure of a forest canopy is significantly impacted either due to harvesting or due to natural disturbance (Coppin et al. 2004). In contrast, it is much harder to distinguish old-growth forests from other forests, because the spectral difference between the two is subtle (Wulder 1998). Consequently, mapping old-growth disturbances based on satellite imagery is thoroughly feasible in areas where an accurate map of old-growth forest distribution is already existent.

Our goal here was to quantify disturbance (defined in our case as full canopy removal due to either natural disturbances, such as wind or insects, or anthropogenic disturbances, such as logging) in Romanian old-growth forests based on the delineations from the last inventory. We recognize that low to moderate intensity wind disturbances and other natural mortality events result in only partial canopy disturbance, with abundant residual live and dead trees (Splechtna et al. 2005; Nagel et al. 2006). In primary systems and where salvage logging does not occur, these biological legacies are incorporated into recovering forests,

often producing multi-aged stands containing remnant old-growth structure (Franklin et al. 2000; Keeton et al. 2010). However, for the purpose of our study we were most concerned with the combined effects of deliberate forest clearing by people and high-intensity (sometimes termed “catastrophic”) natural disturbances. Specifically, we ask the following research questions:

- To what extent did disturbances occur in Romanian old-growth forests between 2000 and 2010?
- How were disturbances distributed among forest-ecozones, and along elevational as well as slope steepness gradients?
- How effective were protected areas in safeguarding old-growth forests in Romania?

2 Study area

The Carpathians are Europe’s largest mountain range, encompassing the continent’s largest continuous temperate forest ecosystem (UNEP 2007). About half of the Carpathian forests are located in Romania. Our study area comprised all of the Romanian Carpathian forests, focusing in particular on old-growth forests (Figure IV-1). Elevation in the study region ranges from 0 m to >2500 m above sea level (a.s.l.) and climate is transitional temperate-continental. Potential natural vegetation of the Carpathian chain generally occurs in altitudinal layers (Donita et al. 1993; Donita et al. 2005). Starting from low to high altitudes, the first layer consists of deciduous forests, with two sub-layers: a pure Sessile oak (*Quercus petraea*) and mixed Sessile oak-European beech (*Fagus sylvatica*) sub-layer (between 300 [400] – 500 [650] m); and a pure beech and mixed beech with silver fir (*Abies alba*) and/or Norway spruce (*Picea abies*) sub-layer (between 500 [650] – 1300 [1450] m) (numbers in squared brackets refer to the Southern Carpathians, otherwise to the Eastern Carpathians). The second layer consists of Norway spruce, with two sub-layers: a mountainous Norway spruce sub-layer (between 1300 [1450] – 1600 [1700] m); and a pre-subalpine Norway spruce sub-layer (between 1600 [1700] – 1750 [1850] m). Above the timber line, there is a sub-alpine layer (1750 [1850] – 2000 [2200] m) with dwarf pine (*Pinus mugo*) and juniper (*Juniperus communis ssp. nana*), followed by an alpine belt (>2000 [2200] m), dominated by dwarf shrubs and short-grass meadows (Cristea 1993; Muica and Popova-Cucu 1993; Feurdean et al. 2007).

The 2001-2004 old-growth forest inventory identified 3402 sites of old-growth forest larger than 50 ha. These old-growth forests were located mainly in montane areas (92% above 600 m) and predominately within the Carpathian Ecoregion (Figure IV-1) (Anfodillo et al. 2008; Veen et al. 2010). European beech is the prevailing old-growth forest type, followed by coniferous forests including Norway spruce, silver fir, Swiss stone pine (*Pinus cembra*) European larch (*Larix decidua*), and Scots pine (*Pinus sylvestris*) (dwarf pine habitats were not included in the assessment) (Biriş and Veen 2005).

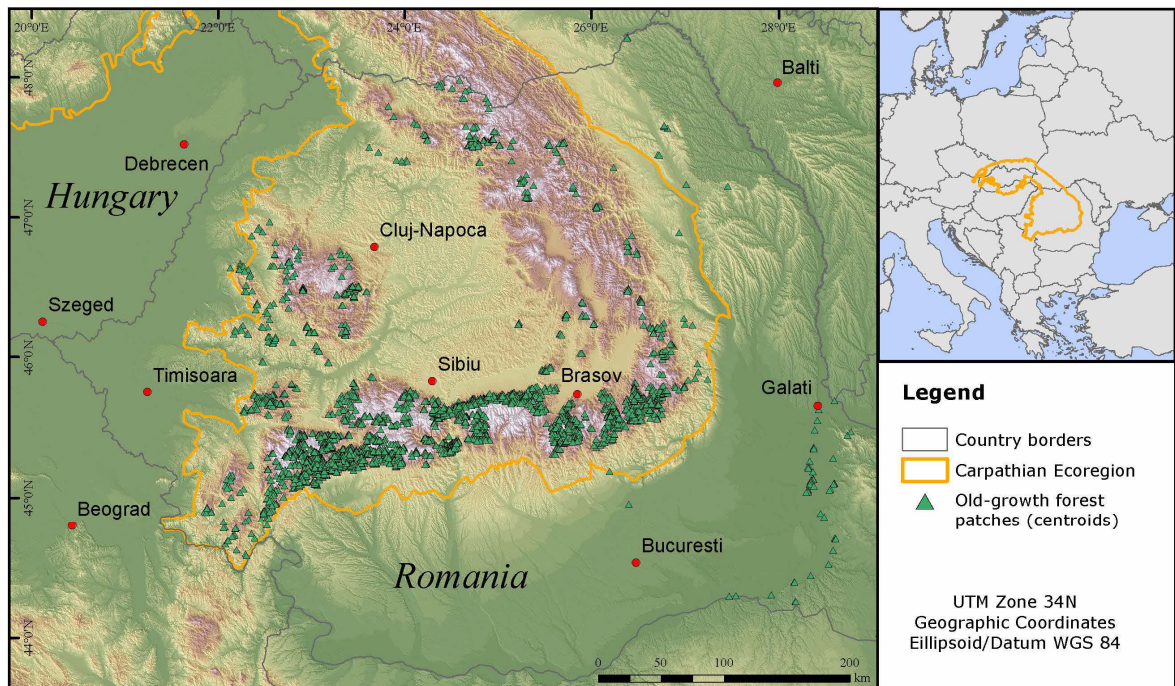


Figure IV-1: Study area in the Carpathian Mountains in Romania including the distribution of old-growth forest patches (Data: SRTM digital elevation model, ESRI Data and Maps Kit).

3 Materials and methods

3.1 Datasets used

We obtained the digital inventory map on areas of old-growth forests (polygon layer) in Romania recorded between 2001-2004 (Biriş and Veen 2005) from the Romanian Forest Research and Management Institute (ICAS) and used it as our baseline.

Forest cover changes from 2000-2010 were mapped across Romania using Landsat TM/ETM+ images (thermal bands were not incorporated) for 16 footprints and with a spatial resolution of either 28.5 or 30 m. Whereas most of the images had already been orthorectified by the United States Geological Survey, several uncorrected images needed

to be co-registered to the others. To do so, about 1500 tie points were located using an automated point matching tool (Leica Geosystems 2006) considering both the acquisition geometry and relief (Griffiths et al. 2012). Results showed an overall positional error below 0.5 pixels.

Additional spatial data included administrative boundaries (ESRI Data and Maps Kit 2008), protected area boundaries including Natura 2000 sites (ICAS), forest-ecozones (ICAS) (Figure IV-2) as well as an enhanced digital elevation model (DEM) based on the Shuttle Topography Mission (SRTM, <http://srtm.csi.cgiar.org>) resampled from 90 to 30 m. Extensive field visits in northern, central-eastern and south-western Romania and interviews with park administrations, stakeholders, NGOs as well as several researchers and other partners were carried out in 2008, 2009, and 2010.

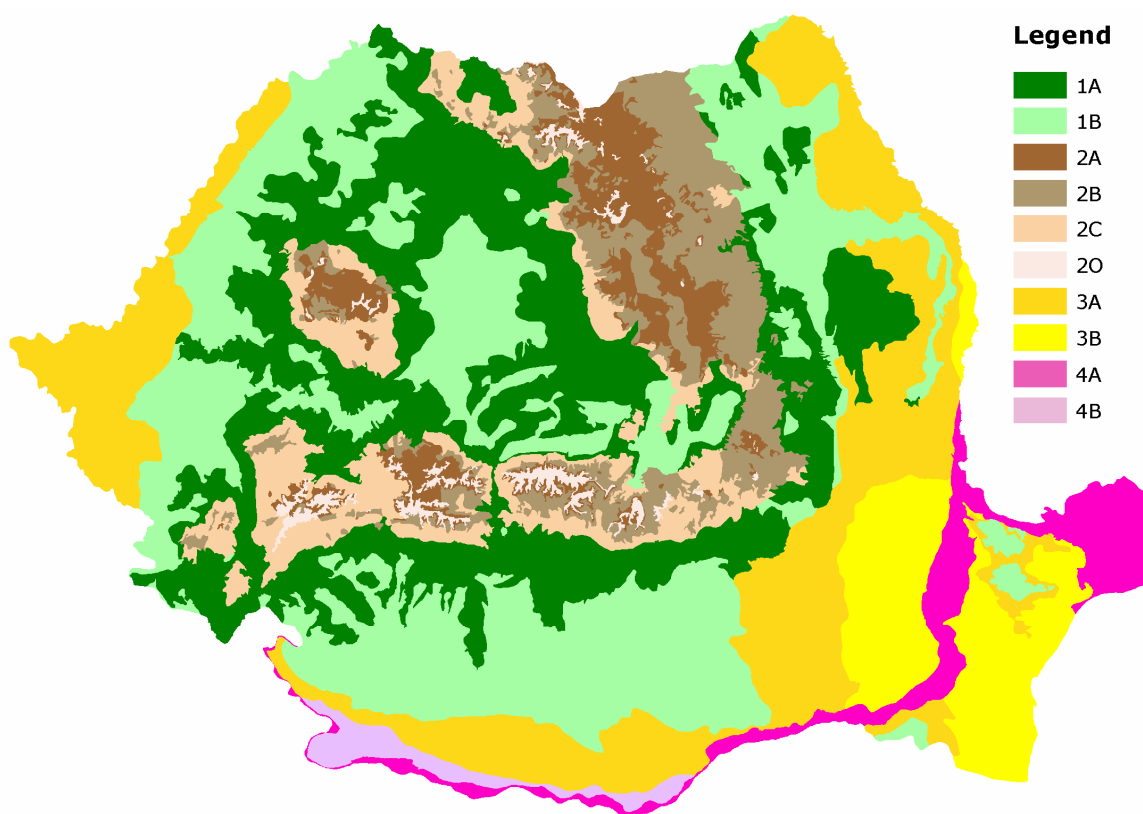


Figure IV-2: Map of Romania's forest-ecozones. 1A = beech and sessile oak mixed forests, Hungarian oak (*Quercus frainetto*) and mixtures, on high and medium hills; 1B = forests with pedunculate oak (*Quercus robur*), Turkey oak (*Quercus cerris*), Hungarian oak and other species, on low hills and plains; 2A = spruce forests; 2B = coniferous and beech mixed forests; 2C = beech mountainous forests; 2O = alpine grasslands and/or bare rocks; 3A = xerophyte oak forests in silvosteppe; 3B = steppe (no natural forest vegetation); 4A = floodplain forests with poplar (*populus*), willow (*Salix*), alder (*Alnus*) and some pedunculate oak; 4B = high floodplain forests with pedunculate oak, ash (*Fraxinus excelsior*) and other.

3.2 Forest disturbance mapping

Forest disturbance maps were obtained from three previous studies with foci on different regions in Romania. The first study by Griffiths et al. (2012) focused on central-eastern Romania (Landsat footprint path/row 183/028) and assessed forest disturbances on an annual basis between 1984 and 2010. The second study by Knorn et al. (in press) analyzed forest disturbances in northern Romania (path/row 185/27) between 1987 and 2010. The third study by Olofsson et al. (2011) assessed forest disturbances between 1990 and 2010 for all of Romania. Based on those maps, an area wide map representing forest disturbances from 2000 to 2010 was assembled. Incorporating each of the studies was necessary since the map by Olofsson et al. partly missed data due to cloud coverage. All three maps were generated using either Support Vector Machines (Pal and Mather 2005), Disturbance Index (Healey et al. 2005), the LandTrendr (Landsat-based detection of trends in disturbance and recovery) set of change detection algorithms (Kennedy et al. 2010), and/or chain classification (Knorn et al. 2009). A detailed description of the specific approaches is found in the original studies of Griffiths et al. (2012), Olofsson et al. (2011) and Knorn et al. (in press). The original forest disturbance maps were subject to individual rigorous accuracy assessments, based on independent ground reference points. Reported overall accuracies of 86.5% (Olofsson et al. 2011), 94.9% (Knorn et al. in press) and 95.7% (Griffiths et al. 2012) yielded prove for the reliability of each map. To build an area-wide forest disturbance map covering all of Romania's old-growth forests, maps from the three original studies were aggregated and the original classes merged to 'permanent forest', 'permanent non-forest', and 'forest disturbance' from 2000 to 2010. While assembling, the maps from Knorn et al. and Griffiths et al. were prioritized due to higher accuracies and temporal resolution. Finally, a minimum mapping unit of ~0.4 ha (i.e., 4 pixels) was applied on the compiled map (Knorn et al. in press).

3.3 Comparison of old-growth forest disturbances

Using the area-wide forest disturbance map, we summarized old-growth forest disturbances based on the polygons from the baseline digital inventory map. We derived proportions of old-growth forest disturbances in relation to both forest ecozones and protected areas. Forest-ecozones delineate Romania's major forest ecosystem regions (Figure IV-2) and were assessed based on existing maps and ancillary data by ICAS-Romania using guidelines from the Joint Research Center (JRC) (Gancz and Pătrășcoiu 2000). For the protected area comparison, we first assessed the total amount of old-growth

forest area and old-growth forest disturbances within protected areas independent of the protection status. Second, we differentiated disturbance rates by protected area types (i.e., Natura 2000, National Park, and Nature Park), but caution that there is a significant overlap between Natura 2000 sites, and other protected areas (Figure IV-3) (Ioja et al. 2010). This means in our case, that about 85% of the National/Nature Park area is simultaneously embedded in Natura 2000 sites. Natura 2000 sites (terrestrial area) cover 42,654 km² corresponding to 17.89% of the Romanian territory and National/Nature Parks 10,800 km² and 4.5%, respectively. Additionally, we assessed the distribution of old-growth forest disturbances in respect to altitude and slope steepness by categorizing the DEM into 100-m wide elevation classes and 8 slope classes each 5° wide.

In addition to the conditions inside old-growth forest patches, the degree of fragmentation on the surrounding forest matrix is also important. Discontinuities and contrast in patch edges enhance the vulnerability of tall, old forests to natural disturbances, alter propagule dispersal, and facilitates movement of invasive species and domesticated fauna (Foster et al. 1996). To determine the intactness of the surrounding landscape, we delineated areas within 250 m of each old-growth forest patch and summarized the forest disturbances within those areas. Areas that were within 250 m of two old-growth forests were counted only once. The area of the buffer zone was equal to the total area of the old-growth forest in the baseline map (around 210,000 ha).

4 Results

In total, 1.3% (about 2720 ha) of old-growth forest was disturbed during the last decade, taking into account that 7238 ha (~3.4%) of the inventoried 210,882 ha old-growth forest stratum could not be classified due to clouds or cloud shadows in the satellite imagery. Old-growth forest disturbances were mainly concentrated along the interior mountain complexes of the Carpathian Ecoregion (Figure IV-3). Clusters of disturbance occurred in the Maramures Mountains in the north, the Apuseni Mountains in the west, and the south/south-western rim of the Carpathian mountain chain (Figure IV-3).

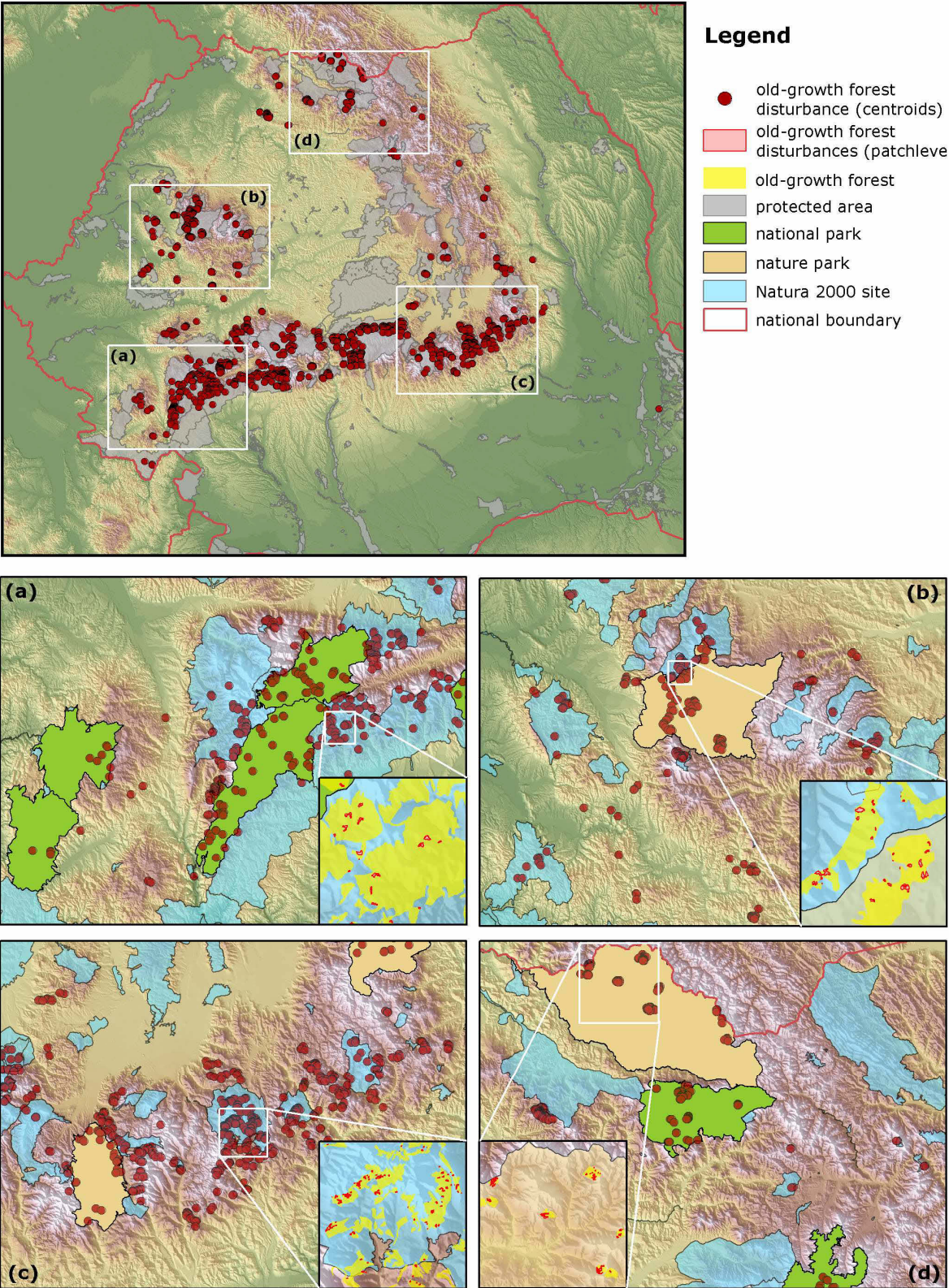


Figure IV-3: Distribution of old-growth forest disturbance patches in Romania. White squares highlight specific areas: (a) South-Western Carpathians, (b) Apunseni Mountains, (c) Curvature Carpathians, and (d) Maramures and Rodna Mountains.

The old-growth forest disturbance map revealed considerable differences in the distribution of disturbance among forest ecozones (Figure IV-4). Disturbances were most prevalent in the forest-ecozone “beech mountainous forests” (about 850 ha), followed by “coniferous and beech mixed forests” (about 726 ha) and “spruce forests” (about 458 ha). Fractions of disturbances among forest ecozones are principally similar to the respective fractions of the original old-growth forest area. However, coniferous forests generally exhibit higher disturbance rates and deciduous forests lower disturbance rates than the respective distribution of old-growth forests would let expect (Figure IV-4).

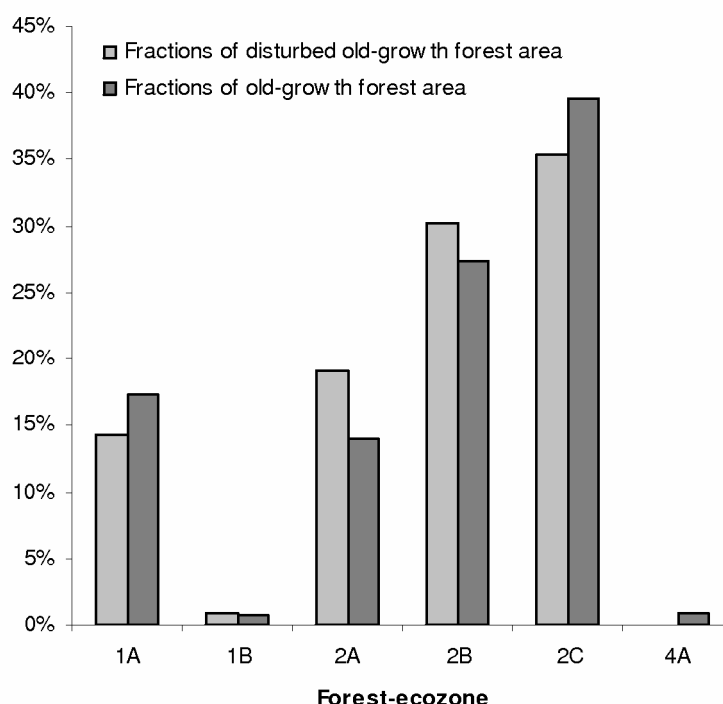


Figure IV-4: Fractions of old-growth forest disturbances and original old-growth forest area in relation to forest-ecozones. Abbreviations for forest-ecozone types are described in Figure IV-2.

The highest amount of old-growth forest disturbances was found at altitudes between 1200 and 1600 m (a.s.l.), whereas their occurrence sharply decreased above 1600 m and gently towards hilly and plain areas below 800 m (Figure IV-5). Fractions of disturbances among altitude were about similar to the respective fractions of the original old-growth forest area (Figure IV-5). Fractions of old-growth forest disturbances among slope gradients aggregate between 15 and 25°, whereas about 6% of all disturbances were found at slopes steeper than 35° (Figure IV-6).

About 77% of the old-growth forest area was embedded within the Romanian protected area network. This included National Parks (23%; 37,917 ha old-growth forests), Nature Parks (14%; 22,435 ha old-growth forests), and Natura 2000 sites (63%; 161,565 ha old-

growth forests, exclusive National/Nature parks) (Figure IV-7). In total, about 72% of all disturbances in old-growth forests were found within a protected area. Of these, 8.5% (167 ha) are in National Parks, 22.1% (432 ha) in Nature Parks, and 69.4% (1359 ha) in Natura 2000 sites (exclusive National/Nature parks) (Figure IV-7).

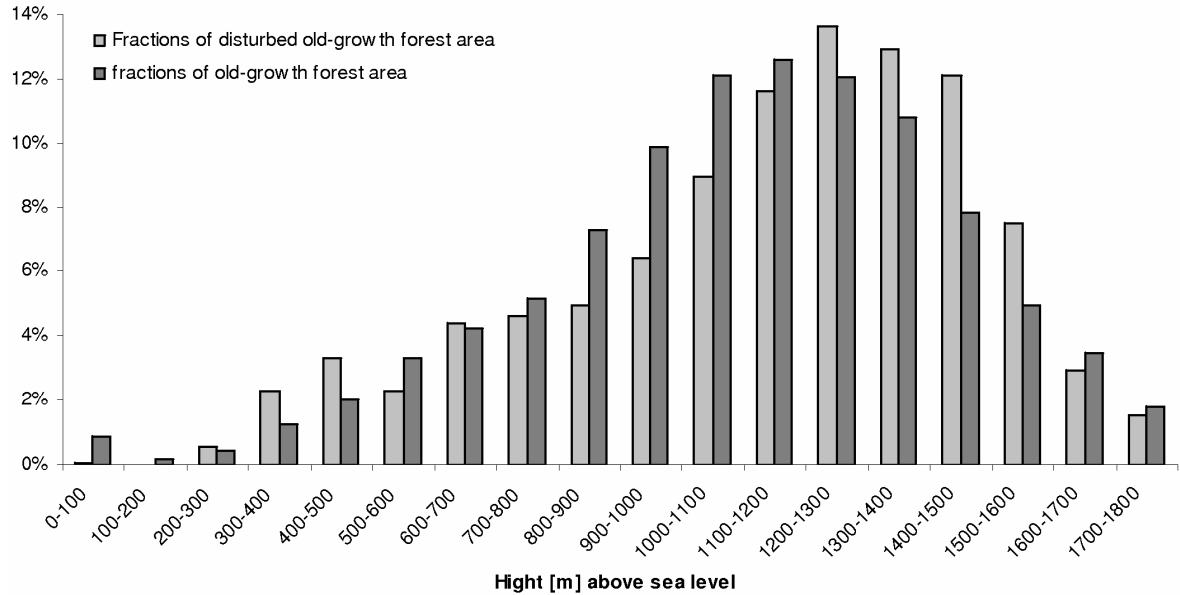


Figure IV-5: Fractions of old-growth forest disturbances and original old-growth forest area in relation to altitude.

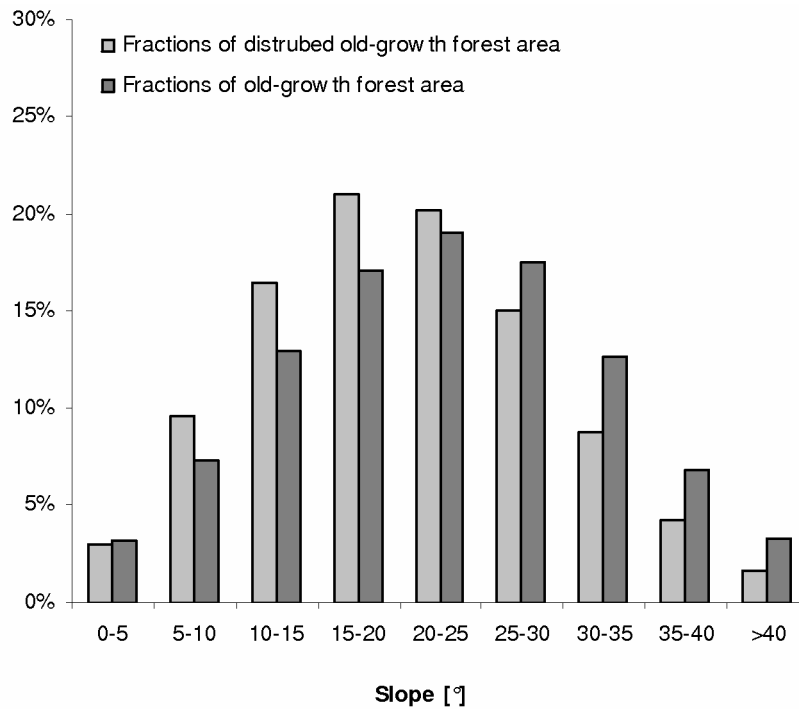


Figure IV-6: Fractions of old-growth forest disturbances and original old-growth forest area in relation to slope.

Disturbances in areas within 250 m of old-growth forests occurred on a total of 3290 ha. This corresponds to about 1.6% of the total buffer area and is thus about 570 ha higher than the respective disturbed area within the original old-growth forest patches.

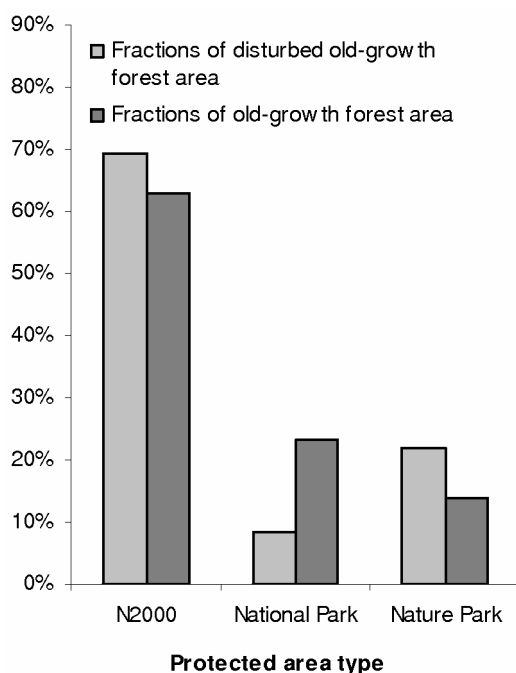


Figure IV-7: Fractions of old-growth forest disturbances and original old-growth forest area in relation to protected areas (N2000 = Natura 2000 site).

5 Discussion

Romania's old-growth forests are threatened by human disturbances. Our remote sensing survey of old-growth forest stands in Romania revealed that disturbances in these stands occurred across the country, but were especially clustered in some areas, for example, in the Apuseni Mountains, the Maramures Mountains, the Curvature Carpathians and the South-Western Carpathians (Figure IV-3). In total, more than 2720 ha of old-growth forests were lost from 2000 to 2010. Although our remote sensing approach could not distinguish between natural and anthropogenic disturbances, yet extensive field visits, interviews with foresters and local experts, as well as our own previous studies (Griffiths et al. 2012; Knorn et al. in press) suggest that natural disturbances alone do not explain this loss. Quite contrary, the observed decline in old-growth forest cover seems to result at least in part from logging. This process is triggered by high-value timber in old-growth stands, institutional changes in the Romanian forest sector, and new ownership structures. Moreover, as cuttings in old-growth forests are predominantly in accordance with forest management plans, legal harvesting activities are obviously responsible for their

diminishing. Additionally, protected areas, including recent expansions under the Natura 2000 framework, additionally appear not as sufficient in safeguarding these forests as they should. Finally, disturbances in the matrix of forest communities surrounding old-growth forest patches additionally affect these old-growth forests negatively (Foster et al. 1996).

Natural stand-replacing forest disturbances, including insect infestation, wind-throw, avalanches and sporadic fires, do occur in the Romanian Carpathian (Schelhaas et al. 2003; Toader and Dumitru 2005). However, most of these disturbance types are either rare or only affect very small areas (i.e., smaller than our minimum mapping unit). Forest fires, for example, are not widespread in the Romanian Carpathians and are a negligible cause of disturbances (Anfodillo et al. 2008; Rozyłowicz et al. 2011). Wind-throw events are both relatively frequent and can cause severe disturbances. This includes our study area and the period observed. Nevertheless, the vast majorities of wind disturbances also affect regions smaller than our minimum mapping unit (Rozyłowicz et al. 2011) and wind-throw events in general show a declining frequency for Romania since 1975 (Popa 2008). Large-scale natural disturbances are also often related to forest management or the legacies from past management (Schelhaas et al. 2003; Mollicone et al. 2006; Schulze et al. 2009). For example, both wind-throws and insect outbreaks are more common in spruce plantations. In the Carpathians these often consist of genetically non-native spruce variants (Keeton and Crow 2009; Kuemmerle et al. 2009; Macovei 2009) which are more susceptible to disease and pests. Likewise, this evidence suggests that natural disturbance events of this type should be significantly larger in areas surrounding old-growth forests which obviously have more substantial forest management histories. Generally, impacts of windstorms on old-growth Norway spruce in the region has been evidenced as well (Panayotov et al. 2011; Svoboda et al. 2012). However, the area affected remained relatively small. Moreover, old-growth spruce forests account only for about 15% of our total old-growth forest area studied. Taken together, natural disturbances are therefore unlikely to explain the majority of forest cover changes in old-growth stands that we observed.

Instead, we suggest that major socio-economic transformations, especially the restitution process, accompanied by institutional changes, considerable economic hardship, and an insufficient protection resulted in logging of old-growth stands, primarily in accordance with forest management plans. Although we caution that a causal connection cannot be established based on our analyses alone, our results, expert interviews, our own previous studies (Griffiths et al. 2012; Knorn et al. in press) as well as extensive field visits unanimously suggest that the observed disturbances are closely related to the forest

restitution process (Irimie and Essmann 2009; Mantescu and Vasile 2009). Widespread clear-cutting was witnessed after the first restitution law in 1991 (Nichiforel and Schanz 2011). Most of the restituted forests (about 300,000 ha) were cleared in the following years by new owners (Mantescu and Vasile 2009). Similar trends occurred in the subsequent restitution phases following the respective laws in 2000 and 2005 (Ioras and Abrudan 2006), all reflected in increased disturbance rates and a higher number of annual disturbance patches compared to the last years of socialism (Griffiths et al. 2012). When the restitution process is finalized, about two-thirds (50% by 2011, according to the Romanian Ministry of Environment and Forests) of Romania's forests will be in private ownership (Ioras and Abrudan 2006). Doubts about the permanence of the newly gained property rights, lack of knowledge regarding sustainable forest management and nature conservation principles (UNDP 2004), as well as the chance to gain short-term profits during times of extreme economic hardship all possibly catalyze the harvesting of restituted forests (Nichiforel 2010; Nichiforel and Schanz 2011). Moreover, structural adjustments necessary to cope with the new ownership structure stay far behind the actual rate of restitution (Irimie and Essmann 2009).

Furthermore, lack of transparency, corruption, and inadequate legal proceedings likely resulted in illegal harvesting activities, even inside protected areas (Brandlmaier and Hirschberger 2005). It has been evidenced, that higher volumes were harvested than official statistics indicate (Bouriaud 2005) and incorrect estimations of wood volume and quality were undertaken (Brandlmaier and Hirschberger 2005). Economic hardship was identified as the main driver of unauthorized logging in the region, as illegal logging was highly correlated with unemployment in rural areas (Bouriaud 2005). Additionally, lack of resources and staff within protected area administrations (Knorn et al. in press) as well as forest inspectorates hamper an sufficient law enforcement. Last but not least, the forest restitution process not only triggered widespread logging in accordance with forest management plans, but also illegal harvesting activities. Accordingly, the intensity of illegal logging and over-harvesting is higher in private forests compared to state forests (Bouriaud 2005).

Most old-growth forest stands and related disturbances were found in mountainous regions dominated by beech forests (Figures IV-2 and IV-3, zones 1A, 2B and 2C), followed by spruce forests (zone 2A). Only very small fractions (0.92%) of disturbances occurred in the foothills or plains (zone 1B), including the Danube floodplain (Figures IV-2 and IV-3, zone 4A), partly because these are areas where few old-growth forests remain. About 50% of all

disturbances occurred at altitudes between 1100 and 1500 m (a.s.l.) (Figure IV-5). Nevertheless, as disturbance fractions correspond to the distribution of the remaining old-growth forest portions, there were only minor deviations in the distribution of disturbances among forest ecozones (slightly more in the coniferous ecozones), altitude (slightly more between 1200 and 1600 m (a.s.l.), or slope (slightly more on slopes with less than 25°). It is conspicuous though, that almost 6% of all old-growth disturbance were found on slopes >35°. Forests on those slopes are protected by law for flood and soil protection (Veen et al. 2010).

Our results also raise concerns about the effectiveness of the protected areas that contain old-growth forests. Nearly 80% of the remaining old-growth forests in Romania are found in protected areas, but 72% of the disturbances happened within their boundaries (Figure IV-7). Moreover, disturbance only differed slightly when comparing rates in protected (1.20%) versus unprotected (1.59%) old-growth forests. Only national parks, which are protected areas with the highest protection status, do indeed effectively prevent disturbances in old-growth forest (Figure IV-7). Several reasons for the apparent ineffectiveness can be named. Although Romania has substantially increased its network of protected areas (Ioja et al. 2010), many still appear to be “protected on paper” only. But most obviously, wood harvesting in old-growth forests is only unexceptionally forbidden inside the core zones of protected areas. Old-growth forests located outside these core areas, but inside buffer areas or completely outside protected areas, are mainly exposed to legal harvesting done in accordance with the forest management plans. The same applies for old-growth forests covered within the Natura 2000 network, where the protection regime allows an active management. Thus, under the precondition of accessibility, harvesting activities in these forests are possible. According to forest management plans, over 13,000 ha of old-growth forests have been included in the functional group of “productive forests” by the date of their last inventory. This basically means that cuttings have been foreseen. Therefore, harvesting in old-growth forests in Romania is beside a few exceptions, in accordance with the law. The only cause that safeguards part of these forests from their potential harvesting is their inaccessibility due to the lack of appropriate infrastructure.

Intensive exploitation of Romania’s forest in the past has generally simplified their composition and forest structure (Toader and Dumitru 2005). This in turn resulted in fragmentation and high-contrast forest edges that increased the vulnerability to wind-throw (Macovei 2009). Fragmentations and potential forest disturbances in the surrounding

matrix of old-growth stands, thus further decreases the possibilities to buffer anthropogenic pressure (Foster et al. 1996).

Our study highlights the fact that intact old-growth forest landscapes continue to disappear in the temperate zone. Romania's protected areas have not been successful in safeguarding these forests, confirming recent concerns about the effectiveness of nature protection in this region (UNEP 2007; Knorn et al. in press). Romania harbours the largest share of Carpathian's high-conservation old-growth forests. If these valuable forests are lost, they cannot be restored in the foreseeable future (Wirth et al. 2009a). This is why institutions at all levels of government need to take action to preserve these last, extensive but vanishing Carpathian old-growth forests. Short-term actions could include a continued monitoring of old-growth stands, and as shown in this analysis, satellite image interpretation offers a promising and valuable tool for doing so. But more important is the protection of old-growth forests against legal cutting (done in accordance with forest management plans). The only effective way for doing so is the strict protection of these forests by law. Consequently, old-growth forests could be incorporated into core protected areas (e.g., IUCN category Ia), given that aims and principles of protected areas are rated higher than the guidelines and regulations of forest management plans. More profound though seems the direct protection through forestry technical provisions in respect to forest management planning. Long-term actions could include the encouragement of communities to equally and sustainably use forest and ecosystem services they provide (Price et al. 2011). Incentives for all private forest owners and especially those holding shares of old-growth forests, should be provided to manage their forests sustainably and compensate them for the loss of opportunities (Brandlmaier and Hirschberger 2005; Dragoi 2010). Last but not least, capacity building and the raise of public awareness (Biriş and Veen 2005) in respect to the exceptional biodiversity and value of ecosystem goods and services old-growth forests provide must be substantially enhanced.

Acknowledgements

We gratefully acknowledge support by the European Union (VOLANTE FP7-ENV-2010-265104), the NASA Land-Use and Land-Cover Change Program of the National Aeronautics and Space Administration (grant number NNX09AK88G), the Belgian Science Policy, Research Program for Earth Observation Stereo II (contract SR/00/133, as

part of the FOMO project), the Czech Science Foundation (grant number GACR P504/10/1644), and the Einstein Foundation.

Chapter V: Synthesis

1 Summary and main findings

Human activities undermine conservation efforts of protected areas worldwide. The first and overarching goal of this thesis was to assess the rates and determining causes of land-use changes in protected areas and forest ecosystems within the Carpathian Ecoregion. Forest cover change was used as the main indicator to prove the effectiveness of protected areas. To that end, a method was first developed to classify large areas in remote sensing data. This method was then used, in conjunction with other techniques, to reveal the major driving forces of land-use change originating from large socio-economic transitions. The study area, located in the Romanian Carpathians, proved to be worth analyzing. Not only due to the fact that more than half and probably the most pristine part of the Carpathian Ecoregion is embedded in Romania, but also because drastic and widespread unsustainable land-cover changes were identified, having profound negative effects on biodiversity and related ecosystem services.

The three main research objectives that were stated in the introductory Chapter I are now reviewed individually:

Research objective 1: To develop a simple, robust, and reproducible method for large area land-cover classification with minimal requirements for image pre-processing and training data.

In Chapter II, a new classification technique for land-cover of large areas was developed. The core idea behind this novel approach was to classify one satellite image first where good ground truth data is available, and then subsequently to classify the neighboring image using the land-cover classification of the first scene in the available overlap area. Using *Landsat* data, this is possible along track as well as across track, incurring only a small loss in accuracy. This approach was called *chain classification* and has been successfully tested for forest/non-forest classification in an 185,000 km² area of the central Carpathian Ecoregion. Note that *chain classification* is not restricted to a particular sensor type, land-cover type, classification technique or geographical region, provided that enough overlap area between scenes exists and land-cover is relative homogenous. *Chain classification* is simple in that it only requires accurate georeferencing of scenes and no atmospheric correction. Accuracy loss between chain classified scenes is 5% or less, even when longer chains are applied. The approach was successfully tested in areas of little or varying training data availability, as it is often the case in the Carpathians (Appendices A

and B). One such study mapped forest cover change and assessed the extent of illegal logging and reforestation in the Ukrainian Carpathians (Appendix A). Therein, Landsat-based forest dynamics differed significantly from official forest statistics. Moreover, widespread illegal logging was revealed even inside restricted areas. Another study assessed the implications of forest restitution on the terrestrial carbon balance (Appendix B). Here, *chain classification* was used to map forest cover for the entire Romanian territory. The study revealed a significant effect of the forest restitution on Romania's terrestrial carbon budget. However, Romania currently remains a terrestrial carbon sink, while offsetting about 7.6% of anthropogenic carbon emissions. Both studies underpin the practical usability and successful implementation of the *chain classification* approach.

Research objective 2: To assess the effectiveness of selected Romanian protected areas at preventing unsanctioned logging and investigate the effects of forest restitution on logging rates and patterns.

In Chapter III, satellite image interpretation was used to measure forest disturbance and compare disturbance rates inside and outside protected areas across a 30,000 km² region in northern Romania. The results show an increasing trend of forest loss during the last decades, which is unlikely to be explained by natural disturbances alone. Instead, major institutional and socio-economic changes can be associated with high rates of disturbance during post-socialism which clearly differ from the disturbance rates observed during the last years of socialism. In particular, the ongoing forest restitution process triggered excessive logging, driven by economic hardship, corruption, doubt about the permanence of new property rights, and lack of knowledge on sustainable harvesting principles. Substantial forest loss was identified inside protected areas, even right within core reserve areas, and in the surrounding matrix of protected areas. Forest disturbance inside and around protected areas threatens habitat integrity, fragments the landscape, affects ecosystem functions and processes, hampers species dispersal, and induces edge effects. Therefore, the effectiveness of protected areas in Romania must be questioned. Additional pressures hampering their effectiveness include the prevalent absence of suitable buffer areas, lack of professional management, financing, and scientific support. Finally, a significant awareness deficit in large parts of the public concerning the importance of natural resources and biodiversity, particularly in respect to non-market ecosystem services, makes an effective management of protected areas even more difficult.

Research objective 3: To analyze the extent of old-growth forest disturbances in Romania and the effectiveness of protected areas to safeguard these forests.

In Chapter IV, a Romanian-wide forest disturbance map covering the complete area of old-growth forests of the country was built using the outcomes of previous studies. Analyzing this map, a total of 1.3% (about 2720 ha) of the original old-growth forest was identified to be disturbed during the last decade. Old-growth forest disturbances were mainly concentrated along the mountain chain of the Carpathian Ecoregion and most present in beech mountainous forests and at altitudes between 1200 m and 1600 m. Disturbance clusters were found around areas with high nature conservation value such as the Apuseni Mountains, the Maramures Mountains, the Curvature Carpathians and the South-Western Carpathians. The observed trend cannot be related to natural disturbances alone. Rather, old-growth forest disturbances originated from anthropogenic interventions, triggered by institutional changes, the high value of old-growth timber, and new ownership structures in the Romanian forest sector.

The results additionally raise concerns about the effectiveness and functioning of Romania's protected areas that contain old-growth forests. It is questionable why wood harvesting in old-growth forests is only forbidden *inside* strictly (core) protected areas. Old-growth forests located *outside* these core areas, but inside buffer areas or completely outside protected areas, are predominantly exposed to legal harvesting done in accordance with the forest management plans. The same applies for old-growth forests covered within the Natura 2000 network. Here, the protection regime allows an active management and therefore legal harvesting of these forests. Moreover, according to forest management plans, over 13,000 ha of old-growth forests have been included in the functional group of "productive forests" by the date of their last inventory. This basically means that cuttings have been intended. Therefore, harvesting in old-growth forests in Romania is, but for a few exceptions, in accordance with the local legislation. The only factor potentially safeguarding these forests is in part their relative inaccessibility due to the lack of appropriate infrastructure. This reveals the unquestionable need for renewed and timely protection of all remaining old-growth forests in Romania. They must immediately and unconditionally be protected by explicit legislation, which must be complemented by appropriate actions at all institutional and governmental levels. Otherwise, the irreversible loss of these last Romanian old-growth forests may not be prevented in time.

Which are the general outcomes from this body of research, and what limitations have been identified?

This thesis highlights how broad-scale socio-economic and political driving forces triggered widespread land-use changes in Romania. Both legal and illegal unsustainable use of resources are mainly caused by rapid changes in these forces. For example, the restitution process almost instantly resulted in widespread forest harvesting, often exceeding sustainable limits. Moreover, momentous socio-economic drivers such as poverty, a low public awareness of the important role non-marked ecosystem services play, institutional decay, corruption, and forest restitution all hamper the effectiveness of Romania's protected areas. In consequence, their ability to safeguard intact old-growth forest landscapes and biodiversity in general, is undercut.

The successful development of the *chain classification* approach in order to classify land-cover over large areas was an important technical prerequisite for this work. Hence, the subsequent studies on protected area effectiveness and old-growth forest across large areas in Romania built upon the use of the approach itself or the knowledge gained throughout the development process. Its utility was also verified in other studies (see appendices), however, some limitations were identified. A basic requirement for the successful application of *chain classification* is the existence of relatively homogenous land-cover types across satellite image footprints. Once distinct land-cover classes change abruptly between image-pairs, the resulting training data may no longer describing them sufficiently well. However, it is important to note that this limitation can be overcome by including manually selected complementary training data, better describing the respective land-cover class in the neighbouring image. Beyond that, *chain classification* is restricted to sensors with sufficient overlap areas between image pairs, which are often compromised by the sensor characteristics itself.

The studies on protected area effectiveness and the status of old-growth forests underpin rising concerns about the priority nature protection is given in Romania. Using remote sensing data, the assessments provide objective proof for prevailing widespread and unsustainable forest loss, complemented by insufficient protected area effectiveness. Nevertheless, ascribing most of the observed forest disturbances to anthropogenic origins alone is not trivial. For Romania, spatially explicit data on restituted forests is not available, and the last national forest inventory dates back to 1984. An unambiguous, strictly causal connection between land-use change and socio-economic forces can thus not

be established with full certitude. However, evidence gathered throughout field-visits, expert interviews, other studies and the trends in the data all unanimously suggest that mainly ownership changes, the forest legislation, and economic hardship are responsible for the disturbances that occurred.

What are the implications of this work, and what can be suggested for further action?

As discussed in the literature review, land-use change is one of the most important drivers of the current biodiversity loss (Pereira et al. 2010; Bellard et al. 2012). The main implementation should thus be a stronger support of strategies that aim at stopping and preventing unsustainable land-use changes. This requires coordinated and concentrated efforts in order to safeguard threatened sites and minimize degradation, fragmentation and destruction of habitats (Boyd et al. 2008; Hoffmann et al. 2010). A continued monitoring of such habitats is therefore also needed, and, as shown in this thesis, satellite image analysis offers a promising and invaluable tool for doing so.

Although environmental markets are not a *panacea* to prevent and reverse unsustainable forest loss, people in these regions should be motivated either through regulation and penalty (e.g., emission limits), cap and trade (e.g., carbon markets), direct payments or self-regulation (Kinzig et al. 2011). Starting at the local level, this would include the encouragement of communities to equally and sustainably use forests and ecosystems services they provide. At the national and international level, governments and associations should develop and provide incentives for landowners to manage their forests in a more sustainable way, and to compensate them for any potential loss of opportunities (Price et al. 2011). Moreover, policies which provide irrevocable protection of old-growth forests and stop the unsustainable harvesting in forest landscapes in general need to be put in place at once.

Changes in biodiversity are directly linked to the proper functioning of ecosystems (Duffy 2008). The outcomes of this thesis show how rapid socio-economic developments translate into the loss of valuable habitat. Weakened institutions and inadequate protection-legislations are main drivers of the continued loss of old-growth forests in Romania. Moreover, forest habitat in general is decreasing primarily due to changes in forest legislation and ownership regimes. Short-term economic profits thus undermine the long term benefits from forest associated ecosystem goods and services. An eroding biodiversity at local or regional scale may not seem serious at first, but may lead to a reduction of resilience at larger spatial scales caused by the degradation of ecosystem functionality

(Midgley 2012). While posing a tremendous challenge, it is thus essential and imperative to address the shortfalls in conservation action in its struggle to stop the loss of global biodiversity. One important step in that direction is the development of the Intergovernmental Platform on Biodiversity and Ecosystem Services (Perrings et al. 2010). Aiming at reducing gaps in the science-policy interface in respect to biodiversity and ecosystem services (Midgley 2012), it could provide the impetus to overcome such challenges (Bellard et al. 2012).

2 Future research

Over the course of this thesis, a number of contributions have been made to large area classifications with remote sensing data on the one hand, and the status and functioning of Carpathian ecosystems on the other. By discussing these important topics, several exciting new ideas evolved and should be investigated in future research.

Assessing post-socialist forest cover change for Romania led to important conclusions about how ecosystems changed in parts of the Carpathian Ecoregion since 1990. If a similar approach was taken for the complete Ecoregion, one could determine further trends in land-use change between countries, investigate habitat fragmentation across borders, compare cross-national protected area effectiveness, and describe the entirety of Carpathians old-growth forests. Official statistics and spatially explicit information on restituted land and logging is available for most of the Carpathian countries. For Romanian, a revised area-wide forest inventory is foreseen to be finalized and available by the end of 2012. Aggregating these statistics and comparing it with forest cover change maps assessed from remote sensing images, might allow the discovery of discrepancies between these two data sources. As shown in other studies from the region (see Appendix A), this is a feasible way to identify for instance illegal logging in the region.

While this study quantified land-cover changes and qualitatively identified underlying drivers, a quantitative assessment of possible causes for land-use change would substantially broaden the understanding of ecosystem-relevant processes in the region. A number of promising approaches have emerged over the course of this study, but a more detailed investigation would be beyond the scope of this thesis. These empirical environmental analyses could include the incorporation of land-use simulations, econometric methods, and coupled human-natural system modelling approaches, thus addressing the effects of policy on both social and natural systems (Butsic et al. 2010).

From a technical perspective, several findings bear potential for further elaborations. *Chain classification* is ultimately limited in its ability to classify very large areas up to a certain size. Even though a chain of up to six images shows only a relatively small loss in accuracy, an area as large as the Carpathians seems not to be continuously classifiable in this way. A promising methodological alternative is the recently emerged *Pixel-Based Compositing Algorithm* (Potapov et al. 2011). Its aim is to produce large area, radiometrically and phenologically consistent cloud-free imagery. The algorithm uses the full *Landsat* archive and extracts the best-suited observation for a specific application on a per-pixel basis. Composite imagery would allow characterizing land-cover uniformly across areas as of the size of the Carpathians. Employing this technique for more than just a single point in time, classification of land-cover changes consistently for the Carpathians appears feasible. This could hence lay the technical foundation to further the above suggested research.

Trajectory-Based Change Detection is another recently developed approach, implemented to better understand processes which drive land-use change in more detail (Kennedy et al. 2010; Griffiths et al. 2012). This method takes advantage of the temporal depths of the *Landsat* archive and uses it to reconstruct detailed histories of land-use and land-cover changes. Characterizing the fine-scaled dynamics of change could help to differentiate, e.g., natural from anthropogenic disturbances. Finally, combining pixel-based compositing and trajectory-based change detection would be a big step forward towards assessing land-cover changes with sufficient detail and accuracy in both space and time.

3 Conclusions

This study broadened the picture about drivers responsible for land-use change in Eastern Europe. Many ecosystems in this region remain threatened, and invaluable habitat continues to be lost at an alarming rate. It is thus of utmost importance to continue the monitoring of land-use and land-cover changes across the Carpathians. This study showed that satellite image interpretation provides an indispensable tool to achieve this goal. But in the end, it is of key importance that these findings are communicated effectively and continuously to policy-makers, donors, and the general public so that appropriate action can be taken. Moreover, capacity building in the respective countries, supplemented by funding strategies fostering holistic research and management strategies are strongly demanded (Price et al. 2011). Yet, ultimately, it is the human society that needs ecosystems

to provide multifaceted services effectively. Any related increase of pressure on ecosystems may not show apparent impacts at first, but will disclose adverse environmental effects after a certain delay, and at a hard-to-predict scale (Costanza et al. 2007). This accounts for local impacts such as unsustainable harvesting up to global impacts such as climate change (Maestre et al. 2012).

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Appendix A:
Forest cover change and illegal logging in the
Ukrainian Carpathians in the transition
period from 1988 to 2007

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Abstract

Illegal logging is a major environmental and economic problem, and exceeds in some countries the amounts of legally harvested timber. In Eastern Europe and the former Soviet Union, illegal logging increased and reforestation on abandoned farmland was widespread after the breakdown of socialism, and the region's forest cover trends remain overall largely unclear. Our goal here was to map forest cover change and to assess the extent of illegal logging and reforestation in the Ukrainian Carpathians. We used Landsat TM/ETM+ images and Support Vector Machines (SVM) to derive forest change trajectories between 1988 and 2007 for the entire Ukrainian Carpathians. We calculated logging and reforestation rates, and compared Landsat-based forest trends to official statistics and inventory maps. Our classification resulted in reliable forest/non-forest maps (overall accuracies between 97.1%-98.01%) and high clear cut detection rates (on average 89.4%). Forest cover change was widespread in the Ukrainian Carpathians between 1988 and 2007. We found forest cover increase in peripheral areas, forest loss in the interior Carpathians, and increased logging in remote areas. Overall, our results suggest that unsustainable forest use from socialist times likely persisted in the post-socialist period, resulting in a continued loss of older forests and forest fragmentation. Landsat-based forest trends differed substantially from official forest resource statistics. Illegal logging appears to have been at least as extensive as documented logging during the early 1990s and so-called sanitary clear-cuts represent a major loophole for overharvesting and logging in restricted areas. Reforestation and illegal logging are frequently not accounted for in forest resource statistics, highlighting limitations of these data. Combating illegal logging and transitioning towards sustainable forestry requires better monitoring and up-to-date accounting of forest resources, in the Carpathians and elsewhere in Eastern Europe, and remote sensing can be a key technology to achieve these goals.

1 Introduction

Changes in forest cover have widespread effects on the provision of ecosystem services, affect biodiversity, and provide important feedbacks to climate change and human welfare (MA 2005; Bonan 2008). As human pressure on the planet rises, monitoring forest cover trends from global to regional scales is therefore of growing international concern (Lepers et al. 2005; Hansen et al. 2008). Official forest resource statistics such as national inventories or the periodic Forest Resource Assessments (FRA) of the Food and Agriculture Organization of the United Nations (FAO 2005) are the most frequently used datasets to monitor forest trends.

The problem is that forest resource statistics often have uneven quality in time and space, inconsistent survey methods, and utilize varying definitions across nations (Rudel et al. 2005; Grainger 2008). Furthermore, official forest resource statistics frequently fail to capture illegal logging, particularly in developing nations, where illegal logging can exceed legal harvesting (WWF 2002, 2004; e.g., > 80% in Indonesia, >50% in Central Africa, or >60% in the Brazilian Amazon, Greenpeace 2008). Assessing the reliability of official forestry statistics and the nature of forest cover trends therefore continues to be a major challenge in many parts of the world and remote sensing plays an important role by providing better estimates.

Illegal logging (i.e., timber harvesting in violation of national laws) can take many different forms and there is no internationally accepted definition of what is illegal (FERN 2002). Two broad categories of illegal logging are usually distinguished (Bouriaud 2005). Timber thefts mostly satisfy local people's demands (e.g., for fuel wood) and are often driven by poverty. On the other hand, unauthorized logging represents timber harvests that deliberately exceed harvesting limits, using corrupt means to gain access to forests, disobeying protected areas and forest laws, or capitalizing on gaps in legislation (Bouriaud and Niskanen 2003; Brack 2007). Thus, unauthorized logging is often connected to failures in forest governance, weak institutions, or a lack of law enforcement (Morozov 2000; Contreras-Hermosilla 2002; Irland 2008). In this article, we consider logging illegal if it is not consistent with harvesting policies and forest laws for any of the above reasons, and therefore not accounted for in official forest resource statistics and inventory data.

Eastern Europe and the former Soviet Union experienced fundamental changes in their political, economic, and institutional structures after the breakdown of socialism. This

raised considerable concerns about forest governance and illegal logging, because the transition period was characterized by economic hardships, and weakened institutions (Kissling-Naf and Bisang 2001; Lerman et al. 2004; Elbakidze and Angelstam 2007). Shadow businesses and corruption in the forestry sector have thrived in some countries, and illegal logging has been reported for the Russian Far East (WWF 2002), Siberia (Vandergert and Newell 2003), northern Russian Karelia (Piipponen 1999), Estonia (Hain and Aha 2004), the Caucasus (Greenpeace 2000), and the Carpathian Mountains (Turnock 2002). Substantial proportions of timber exports from Eastern Europe and European Russia are illegal (Bouriaud and Niskanen 2003; WWF 2004). However, the extent of illegal logging remains unclear, and available estimates vary among different sources (Bouriaud 2005).

The transition from planned to market-oriented economies in Eastern Europe also resulted in widespread farmland abandonment particularly on marginal sites (Ioffe et al. 2004; DLG 2005; Kuemmerle et al. 2008). Much of the abandoned land is now reverting back to forests, but just like illegal logging, reforestation (i.e., forest expansion via natural succession or planting) is frequently not included in official forest statistics. This impedes assessing net forest cover changes in Eastern Europe and the former Soviet Union, hampers subsequent analyses such as carbon budgeting, and poses serious challenges for policy makers aiming to implement sustainable forest management plans.

Eastern Europe is also still rich in vast and relatively undisturbed forest landscapes (Wesolowski 2005). For example, the Carpathian Mountains constitute Europe's largest temperate forest ecosystems and are a biodiversity hotspot (UNEP 2007). The Ukrainian region of the Carpathians is particularly important, because it bridges the northern and southern Carpathians, and includes some of Europe's last and largest old-growth beech forests (Herenchuk 1968; Holubets et al. 1988; Wesolowski 2005). Forest use has changed substantially in the Ukrainian Carpathians after the country became independent in 1991. Forest harvesting increased in some areas (Kuemmerle et al. 2007) and illegal logging occurred (Nijnik and Van Kooten 2000; Buksha et al. 2003). On the other hand, forest expansion on abandoned farmland was widespread (Elbakidze and Angelstam 2007; Kuemmerle et al. 2008) and Ukraine issued a national forest planting program in 2002. These opposite processes raise questions about net forest cover trends in the Ukrainian Carpathians in the post-socialist period. Unfortunately, available statistical forest resource data provide vastly differing numbers. For example, harvesting rates between 1991-1995 reported by Nilsson and Shvidenko (1999) are up to 60% higher than Zibtsev's (1998)

rates. Even the direction of post-socialist harvesting trends is unclear with most studies reporting decreased harvesting (Nilsson and Shvidenko 1999; Buksha et al. 2003; FAO 2005), whereas others suggest increased logging during the early 1990s (Nijnik and Van Kooten 2000). Overall, net forest cover changes in the Ukrainian Carpathians since the breakdown of socialism have only been examined for small study areas (Kozak et al. 2007b; Kuemmerle et al. 2007; Sitko and Troll 2008) and no study has so far compared actual forest cover change with official forest resource data.

The lack of an area-wide forest change map is partly explained by the challenges that large-area mapping of forest cover change in mountain regions face. Phenology, illumination effects, and variability in vegetation communities along altitudinal gradients frequently result in spectrally complex thematic classes (i.e., multi-modal, non-normal, Itten and Meyer 1993; Seto and Liu 2003). Non-parametric classifiers are powerful tools in such situations, because they do not assume specific a-priori density distributions per class (Friedl and Brodley 1997; Seto and Liu 2003). Support Vector Machines (SVM) perform equally well or better than other non-parametric approaches, while requiring fewer training samples (Foody and Mathur 2004b; Pal and Mather 2005). SVM discriminate classes by fitting a separating hyperplane in the feature space based on training samples (Huang et al. 2002) and have been successfully applied to map forest cover changes over large areas (Huang et al. 2008; Kuemmerle et al. 2008).

The increasing availability of long image time series, such as the Landsat data archive, now allows for moving from simplistic from-to assessments towards detailed change trajectory analyses (Hostert et al. 2003; Kennedy et al. 2007; Röder et al. 2008). Many change detection methods exist to analyze image pairs (Coppin et al. 2004), but tools for investigating dense time series of Landsat imagery are largely lacking (Kennedy et al. 2007). The challenge is that the complexity of a composite classification (Coppin and Bauer 1996) increases exponentially with every additional image (e.g., 256 change classes for four land-cover classes and four time periods). This often inhibits the collection of a representative training sample. In such situations, classifying images individually and assessing change a-posteriori may be the better option, if accurate individual classifications can be achieved.

Our goal here was to assess the extent of illegal logging and reforestation in the Ukrainian Carpathians by exploring whether post-socialist forest cover trends mapped from satellite images differed from those reported in official forest resource data and forest inventories.

This required us to derive the first area-wide forest cover change map for the Ukrainian Carpathians using Landsat Thematic Mapper (TM) and Enhanced Thematic Mapper Plus (ETM+) images. Our specific objectives were to:

- map forest cover change in the Ukrainian Carpathians before and after the system change (1988 – 2007) from Landsat TM/ETM+ images using SVM,
- compare satellite-based forest trends with official forest statistics,
- compare satellite-based logging maps with forest inventory maps, and
- assess the spatial pattern of logging in relation to topography and the visibility of logging sites.

2 Study region

As a study region, we selected the entire Ukrainian Carpathians (Figure A-1). Study region boundaries were based on administrative borders at the county (raion) level and we selected all raions that were at least partly within the ecoregion (using the Carpathian Ecoregion Initiative's boundary, www.carpates.org, Kruhlov 2008). The study region intersects with four provinces (oblasts): Chernivetska Oblast, Ivano-Frankivska Oblast, Lvivska Oblast, and Zakarpatska Oblast. It covers an area of 30,890 km², and its total population is about 2 million (UNEP 2007).

The region is characterized by a northwest-southeast running mountain range, predominately consisting of flysch and some volcanic and metamorphic rocks in the southwest. Altitude varies from >100 m to 2061 m. The climate is temperate with a moderate continental influence and varies significantly depending on topography (Herenchuk 1968; temperature range between 20° C to 6° C in summer and -3° C to -10° C in winter; annual precipitation is 900–1200 mm, Buchinskyi et al. 1971). Four altitudinal zones of natural vegetation occur in the study region. The foothills and adjacent plains (<300 m) are covered by broadleaved forests with pedunculate and sessile oak (*Quercus robur*, *Q. petraea*), often mixed with European beech (*Fagus sylvatica*), linden (*Tilia cordata*), hornbeam (*Carpinus betulus*), and ash (*Fraxinus excelsior*). The lower montane zone (300-1100 m) consists of beech forests with silver fir (*Abies alba*), Norway spruce (*Picea abies*), and sycamore maple (*Acer pseudoplatanus*). The upper montane zone extends up to the timberline (1500 m) and is dominated by coniferous species, mainly spruce and Arolla pine (*Pinus cembra*). Above timberline, mountain pine (*Pinus mugo*),

green alder (*Alnus viridis*), and juniper (*Juniperus communis subsp. alpina*) shrubs and alpine grasslands prevail (Herenchuk 1968; Kruhlov et al. 2008).

Land-use has substantially affected the Ukrainian Carpathians. Much of the region's forestland was converted to farmland during the Austro-Hungarian Empire (1772-1918) and the foothill zone remains dominated by agriculture. Since the early 20th century, forest cover has been increasing slowly (Kozak et al. 2007a). However, forests have also been excessively exploited since the 19th century, especially under Soviet rule, resulting in an age distribution dominated by young age classes, increased forest fragmentation, and widespread spruce plantations (Strochinskii et al. 2001; Turnock 2002; Irland and Kremenetska 2008). Mountain tops have traditionally been used for grazing, resulting in lowered timberlines in some regions (Sitko and Troll 2008).



Figure A-1: Study region in the Ukrainian Carpathians. Main frame: study region boundaries (red), topography (elevation range >100-2060 m), and major population centers. Inset A: location of the study region in Europe. Inset B: the four provinces (oblasts) comprising the Ukrainian Carpathians. Inset C: Major roads and railway tracks. Source: SRTM DEM (elevation data); ESRI World Data and Maps Kit 2005 (national boundaries and population centers); Geodezkartinformatyka 1997 (oblast boundaries, roads, railways).

3 Datasets used and methodology

3.1 Satellite images and ancillary data

We acquired 19 mid-summer and early fall Landsat TM and ETM+ images for ~1988, 1994, ~2000, and ~2007. Five Landsat footprints covered the full extent of the study region (path/row 184/26, 184/27, 185/26, 185/27, and 186/26). Full cloud-free coverage of the study region for a single year was only possible for 1994. For the late 1980s and the most recent time period (2007) we used images acquired ± 1 year, and two 2002 images complemented the 2000 imagery (acquisition dates are listed in Table A-2 in the results section).

Five of the images from the GeoCover dataset were already orthorectified (Tucker et al. 2004). The remaining 14 images were co-registered to the GeoCover dataset. To account for relief displacement, we included the Space Shuttle Topography Mission (SRTM) digital elevation model, resampled to 30 m. Tie points between GeoCover and uncorrected images were gathered automatically based on image correlation (Kuemmerle et al. 2006). All co-registered images had a positional accuracy of <0.5 pixel (Tucker et al. 2004). Clouds and cloud shadows were digitized and masked.

Ground truth data were gathered based on approximately 120 Quickbird images from 2002 to 2007 available in Google Earth™ (earth.google.com), covering 43.3% of our study area. Overlaying topographic maps and GPS tracks gathered in the field between 2004 and 2006 suggested that the positional accuracy of the Quickbird images was comparable to that of the Landsat images. For each Landsat footprint, we selected a random sample of ground truth points within the Quickbird image footprints, overlaid points on the Quickbird images in Google Earth™, and labeled each point as either 'forest' or 'non-forest' based on visual interpretation. A point was considered forested if tree cover exceeded 60% (i.e., 'closed tree cover' in the Land Cover Classification System, Di Gregorio 2005) and if tree-dominated patches covered at least one Landsat pixel (30x30 m). Thus, our forest definition included orchards, but not single trees, treelines, and open shrubland. We only considered points where class membership was stable between 1988-2007 (i.e. either permanent forest or permanent non-forest), based on visual interpretation of the Landsat images. Ground truth points with unclear class membership, points in cloud areas, and points closer to forest/non-forest borders than the remaining positional uncertainty (less than 15 m) were

discarded (3% of a random sample at most). Training samples of 300 to 500 ground truth points per class resulted in stable classification accuracies and 1400 random points per image provided this minimum amount of points per class (Knorn et al. 2009). For three footprints (path/row 185/26, 185/27, and 186/26), we also used ground truth points mapped in the field between 2004 and 2006 (Kuemmerle et al. 2007; Kuemmerle et al. 2008). Points in overlap areas between Landsat paths were used for both footprints. In total, we used a sample of 5211 points (2373 forest, and 2838 non-forest) of which 4481 (1976, 2505) were mapped from Quickbird data and 730 (397, 333) were mapped in the field (see section 4, Table A-2).

Administrative boundaries at the province (oblast) and district (raion) level were digitized from topographic maps at a scale of 1:100,000 while road and railway networks were extracted from the digital topographic maps at a scale of 1:200,000 (Geodezkartinformatyka 1997). Roads were classified as highways, paved roads, or dirt roads, and railway tracks as major tracks or narrow-gauge tracks. Country boundaries and major population centers were obtained from the Environmental Systems Research Institute's (ESRI) World Data & Maps Kit 2005.

Forest resource statistics at the oblast level were obtained from the Statistical Yearbook of Ukraine for the years 1985-1987, 1990, 1995, 2000, and 2002-2007 (The State Statistics Committee of Ukraine 2006, 2007). We extracted two indicators: (1) areas designated for post-clear-cut forest regeneration (i.e., where forest regeneration should have been carried out), and (2) areas dedicated for new forest planting. We also acquired a digital forest inventory map at a scale of 1:10,000 covering all state-managed forests in Zakarpatska Oblast. This map contains more than 89,000 polygons and provided detailed stand-level information on forest management practices carried out between 1999-2007. Polygons in this map represent the finest scale of forest management units in the Ukrainian Carpathians. We categorized all forest management practices into practices where forest cover is retained, partly removed (e.g. single or group selection harvesting), or fully removed (e.g., clear-cuts). Planned forest management practices that had not yet been implemented were excluded. Where several forest management practices had been carried out (e.g., clear-cutting followed by forest regeneration practices), we considered only the oldest practice where forest cover was fully removed (or partially removed where forest cover was never fully removed). To assess the accuracy of the forest inventory maps, we randomly selected 100 polygons designated as clear-cuts and checked them visually by overlaying them with the Landsat images. Each polygon was assigned to one of the three

classes 'No forest cover removal', 'Partial forest cover removal', or 'Complete forest cover removal'.

3.2 Forest cover change mapping using support vector machines

Image classification with SVM is based on fitting a separating linear hyperplane between two classes in the multidimensional feature space (Huang et al. 2002; Foody and Mathur 2004a). The optimal hyperplane is constructed by maximizing the margin between training samples of opposite classes. Thus, instead of using all available training data to describe classes, SVM use only those training samples that describe class boundaries, the so-called support vectors (Foody and Mathur 2004b, 2006). To separate classes with non-linear boundaries, kernel functions are used to transform training data into a higher-dimensional space, where linear class separation is possible (Huang et al. 2002). This allows SVM to effectively handle complex class distributions (i.e., non-linear, multi-modal) while requiring relatively few training samples (Foody and Mathur 2004b; Pal and Mather 2005). A detailed mathematical description of SVM concepts is found in Burges et al. (1998). Detailed introductions in a remote sensing context are provided by Huang et al. (2002) and Foody and Mathur (2004a).

We used SVMs to delineate forest/non-forest maps for each of the four time periods and assessed forest cover change via post-classification map comparison. This reduced the complexity of our classification approach to a binary problem for which SVM were originally developed (Huang et al. 2002). As a kernel function, we decided to use a Gaussian radial basis function (Huang et al. 2002), that requires setting the kernel width (γ). Parameterizing the SVM also requires setting a regularization parameter C, that penalizes misclassified training data to control the trade-off between maximizing the margin and training error (Pal and Mather 2005). Small C-values tend to emphasize the margin while ignoring outliers, whereas large C-values may result in over-fitting. Thus, the best-performing combination of γ and C depends on the training data and is not known a priori. We systematically tested a wide range of parameter combinations (γ from 0.00001 to 100000 and C from 0.1 to 1000) by fitting individual SVM to each parameter pair and comparing models based on cross-validation errors (Janz et al. 2007; Kuemmerle et al. 2008). This allowed us to identify optimal parameter combinations for each image individually.

Once optimal γ and C were found, we classified each of the 19 Landsat TM/ETM+ images based on the six multi-spectral bands. We split all available ground truth points into

training (90%) and validation (10%) samples. Based on the validation sample, we then calculated an error matrix, overall accuracy, user's and producer's accuracy, and the kappa statistics (Congalton 1991; Foody 2002). We also derived the F-measure, an indicator of overall classification accuracy based on the weighted harmonic mean of producer's and user's accuracy (Baeza-Yates and Ribeiro-Neto 1999). To derive robust error estimations, we classified each image 10 times for all 10 possible splits, derived the accuracy measures, and then calculated mean error estimates (Friedl and Brodley 1997; Steele 2005). The final classification was calculated using 100% of the ground truth data, and the mean error estimate is thus a conservative estimator of the true accuracy (Burman 1989). The SVM parameter search, image classification, and accuracy assessment were carried out with the software imageSVM (www.hu-geomatics.de).

We mosaicked the forest/non-forest maps for each time period. Maps with higher accuracy were given priority in overlap areas and we filled clouded areas with data from overlapping paths wherever possible. Remaining clouds were masked from all mosaics (<1.0% of the study region). Once mosaics for all four time periods were available, we established a rule-set to derive a forest cover change map (Table A-1). Depending on the time of disturbance and the post-disturbance regeneration, we defined eight disturbance classes. The term disturbance here refers to the complete or near-complete removal of forest cover by anthropogenic processes (e.g., logging) or natural events (e.g., storms).

We assumed reforestation on abandoned farmland to take longer than six years, because forest planting virtually stopped after the system change and natural succession is slow in the Carpathians (Buksha et al. 2003; Kuemmerle et al. 2008). Farmland abandonment was not widespread before 1988, and this means that forest regeneration in 1988-1994 largely reflected pre-1988 disturbances, and reforestation could not have occurred before 1994-2000. Initial tests suggested that the reforestation classes contained some disturbances where forest regeneration was slow. We therefore selected all reforestation patches within forests (>80% relative border to permanent forest or disturbances) and assigned such patches to the pre-1988 disturbance class. Four of the possible 16 change classes suggested two disturbance events within a 12-year period (Table A-1). Comparing these classes to high-resolution imagery revealed that they almost exclusively represented misclassifications due to phenology differences among images and all such patches were assigned to 'Permanent non-forest'. These four classes together covered about 1.2% of the study region.

Table A-1: Rule set for delineating the forest cover change map based on the forest/non-forest classifications for each time period (F = forest; NF = non-forest).

ID	Class Label	Time Period			
		1988	1994	2000	2007
1	Permanent forest	F	F	F	F
2	Permanent non-forest	NF	NF	NF	NF
3	Forest disturbance before 1988	NF	F	F	F
4	Forest disturbance in 1988-1994 (a)	F	NF	F	F
5	Forest disturbance in 1988-1994 (b)	F	NF	NF	F
6	Permanent clearing in 1988-1994	F	NF	NF	NF
7	Forest disturbance in 1994-2000 (a)	F	F	NF	F
8	Forest disturbance in 1994-2000 (b)	F	F	NF	NF
9	Forest disturbance in 2000-2007	F	F	F	NF
10	Forest disturbance before 1988 and in 2000-2007	NF	F	F	NF
11	Reforestation 1988-2000 or Forest disturbance before 1988	NF	NF	F	F
12	Reforestation 1988-2000-2007	NF	NF	NF	F
13	Misclassification (a)	NF	F	NF	F
14	Misclassification (b)	F	NF	F	NF
15	Misclassification (c)	NF	NF	F	NF
16	Misclassification (d)	NF	F	NF	NF

To eliminate small disturbance patches representing mostly misclassifications, small forest patches that were functionally not forest (e.g., hedgerows, trees along roads and creeks, groups of trees between fields, etc), and the salt-and-pepper effect common to pixel-based classifications, we assigned patches with <7 pixels to the dominating surrounding class. This threshold was selected because the smallest forest management unit in Ukraine is 0.5 ha. We also selected all disturbance patches fully surrounded by non-forest, disturbance patches (but not reforestation patches) above the timberline (i.e., mean elevation of >1350 m and relative border to permanent non-forest >0.8), and narrow disturbances along rivers (disturbance patches with length/width ratios >4.5) and assigned them to permanent non-forest, because field visits and high-resolution images suggested these patches represented mostly misclassifications.

In addition to the accuracy assessment of the individual classifications, we conducted a validation of the detectability of disturbances. We randomly selected 25 points and digitized the closest disturbance for each of the four time periods based on the Landsat TM/ETM+ images. This resulted in a total of 100 disturbance polygons, together covering an area of 877 ha, and we cross-tabulated these areas with the forest cover change map.

3.3 Analyzing forest cover change

To compare forest change among different regions and time periods, we calculated absolute and relative net forest cover changes as well as annual disturbance and reforestation rates for the full study region, for each oblast, and for each raion. Net change was calculated as the difference in forest cover (in km²) between 1988 and 2007, whereas relative net change (RNC) was calculated as:

$$RNC = (FC_{2007} / FC_{1988} - 1) * 100 \quad (1)$$

where FC denotes forest cover (in km²). Annual disturbance rates (DR) were calculated for each time period *j* as:

$$DR_j = (D_j / FCB_j) * 100 / a \quad (2)$$

where *D* is the sum of disturbances in time period *j*, *FCB* denotes forest cover at the beginning of time period *j*, and *a* is the number of years between image acquisition. Because images from one time period were not always from a single year, we intersected the Landsat footprints from the beginning and end of a time period (considering how images had been mosaicked in overlap areas to adjacent footprints). We then assigned the number of years between image acquisition (*a*) for each segment, and calculated disturbance rates per segment. To summarize disturbance rates at the study region, oblast, and raion level, we calculated the area-weighted mean of disturbance rates. Detection of older disturbances in temperate forest ecosystems can be challenging because of forest regeneration (Healey et al. 2005; Kennedy et al. 2007). We thus decided to use a maximum *a* of 6 years based on prior experience (Kuemmerle et al. 2007). Reforestation rates (RR) were calculated as:

$$RR_j = (R_j / NF_{1988}) * 100 \quad (3)$$

where *R* is the reforestation area per time period, and *NF*₁₉₈₈ denotes all non-forest land (excluding disturbances) in 1988.

To assess whether forest cover change varied with altitude, we stratified the DEM into 100 m strata and calculated mean annual disturbance rates for each stratum and time period. Likewise, we summarized disturbance rates for 9 slope classes using 5-degree breaks. To compare the forest inventory map and the Landsat forest cover change map, we summarized unchanged, disturbed, and reforested areas from the change map for each

forest management practice and for each of the three aggregated forest management categories (full, partial, or no forest cover removal).

New forest legislation prohibiting clear-cuts in beech-fir forest above 1100 m and on steep slopes > 20 degrees was put in place in Ukraine in 2000 (Verkhovna Rada 2000a, b). To assess how these policies affected disturbance rates, we summarized the disturbance area above 1100 m and on slopes steeper than 20 degrees for each time period. Because illegal logging is often hidden, we also assessed the proportion of disturbances visible from highways, paved roads and railway tracks using a viewshed analysis. We categorized our study region into areas that were either visible or invisible from these features and summarized disturbances for both categories and each time period.

4 Results

Our SVM-based classification approach resulted in reliable forest/non-forest maps for all Landsat TM/ETM+ footprints and time periods. Overall accuracies of the individual classifications ranged from 94.68 to 99.40% (kappa 0.88 to 0.98, Table A-2).

Table A-2: Landsat TM/ETM+ images used and classification accuracies [%] of the forest (F) / non-forest (NF) maps for each image. U.A. = User's Accuracy; P.A. = Producer's accuracy.

Time Period	Path/ Row	Acquisition Date	Sensor	Overall Accuracy	Kappa	U.A.		P.A.		F-measure		Number of points		
						F	NF	F	NF	F	NF	Total	F	NF
1988	184/26	1989/07/08	TM5	98.72	0.97	97.87	99.08	97.50	99.18	97.69	99.13	1343	369	974
	184/27	1989/07/08	TM5	95.43	0.90	96.59	93.26	96.59	93.10	96.59	93.18	1279	851	428
	185/26	1988/08/21	TM4	97.94	0.96	97.64	98.20	97.41	98.31	97.52	98.26	1316	543	773
	185/27	1988/08/13	TM5	97.59	0.94	94.56	98.83	97.00	97.82	95.76	98.32	1096	307	789
	186/26	1988/07/27	TM4	98.05	0.96	97.85	98.22	97.00	98.67	97.42	98.45	1342	503	839
1994	184/26	1994/09/08	TM5	98.65	0.97	97.32	99.18	97.78	98.97	97.55	99.07	1343	369	974
	184/27	1994/09/08	TM5	97.34	0.94	98.07	96.01	97.93	96.19	98.00	96.10	1255	826	429
	185/26	1994/07/29	TM5	98.14	0.96	97.49	98.66	98.18	98.11	97.83	98.38	1297	552	745
	186/26	1994/07/04	TM5	97.15	0.94	94.91	98.62	97.76	96.79	96.31	97.70	1309	490	819
2000	184/26	2002/08/21	ETM+	99.40	0.98	98.65	99.69	99.17	99.48	98.91	99.59	1343	369	974
	184/27	2002/07/04	ETM+	96.61	0.92	96.34	97.30	98.61	92.86	97.46	95.03	1220	797	423
	185/26	2000/06/03	ETM+	96.34	0.93	95.89	96.90	96.82	95.86	96.35	96.38	1738	859	879
	185/27	2000/08/22	ETM+	96.55	0.92	93.47	97.98	95.23	97.12	94.34	97.55	1493	448	1045
	186/26	2000/06/10	ETM+	97.04	0.94	95.40	98.17	97.18	96.94	96.28	97.55	2003	785	1218
2007	184/26	2006/09/25	TM5	98.80	0.97	97.07	99.49	98.61	98.87	97.84	99.18	1343	369	974
	184/27	2006/10/11	TM5	94.68	0.88	94.70	94.87	97.70	88.29	96.18	91.46	1103	746	357
	185/26	2007/07/17	TM5	96.59	0.93	93.43	99.17	98.87	95.00	96.07	97.04	1296	531	765
	185/27	2007/07/17	TM5	97.94	0.95	96.30	98.65	96.75	98.44	96.52	98.54	1364	403	961
	186/26	2007/07/24	TM5	95.57	0.91	92.87	97.41	95.71	95.49	94.27	96.44	1311	491	820

Disturbances in 1988-1994, 1994-2000, and 2000-2007 were captured with high accuracies (>87%, Table A-3). Disturbances before 1988 were detected with a slightly lower accuracy (83%), due to confusion with permanent non-forest areas. The overall accuracy of our change map, estimated as the product of the individual map accuracies (Coppin et al. 2004), was 95.81% for the 1988-1994 period, 95.29% for the 1994-2000 period, and 94.61% for the 2000-2007 period.

Table A-3: Validation of disturbance detectability before 1988, in 1988-1994, in 1994-2000, and in 2000-2007. Numbers indicate relative abundance (in percent) of different classes mapped from the Landsat TM/ETM+ images in each of the four disturbance categories of the reference data.

		Reference data			
		Disturbances before 1988	Disturbances in 1988-1994	Disturbances in 1994-2000	Disturbances in 2000-2007
Classified data	Permanent forest	3.02	7.93	8.20	2.29
	Permanent non-forest	8.98	0.82	0.00	0.14
	Forest disturbance before 1988	83.49	0.29	0.00	0.00
	Forest disturbance 1988-1994	2.35	87.67	0.85	0.14
	Forest disturbance 1994-2000	0.14	2.73	89.70	0.70
	Forest disturbance 2000-2007	0.32	0.57	1.25	96.73
	Reforestation 1988-2000	0.14	0.00	0.00	0.00
	Reforestation 1988-2000-2007	1.58	0.00	0.00	0.00

Forest cover changed substantially in the Ukrainian Carpathians between 1988 and 2007 (Figure A-2, top) mainly due to disturbances, which affected 6.83% of the study region (2072 km²). Forests had regenerated on the majority (1365 km²) of these areas in 2007. Disturbances occurred highly clustered. Before 1994 disturbance clusters were mainly found in the northern and southwestern foothills, and close to the Romanian border. After 1994, disturbance clusters mainly occurred in the interior Carpathians. Reforestation occurred on 2.25% of all non-forest land in 1988 (equaling 306 km²), mainly in the plains in the Southwest and Northeast of the study region (Figure A-2, top). Overall, forest cover increase slightly in the Ukrainian Carpathians between 1988-2007 (0.82% of the study region, equaling 250 km²).

Our Landsat-based forest cover change map differed markedly from the forest inventory map (Figure A-2, bottom). Between 2000 and 2007, most disturbances mapped in the satellite images were documented as clear-cuts in the forest inventory maps, although clear-cuts were sometimes larger than documented. Conversely, there were also clear cuts in the inventory data that appeared only partially or not at all harvested in the satellite images. Before 2000, only a relatively small proportion of the disturbances appeared in the

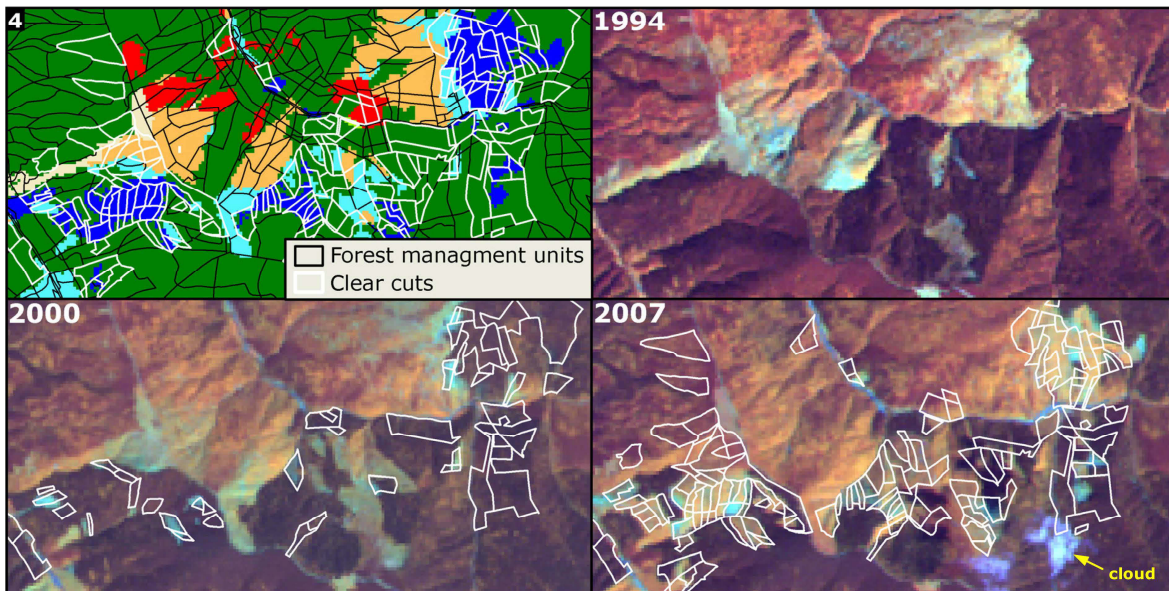
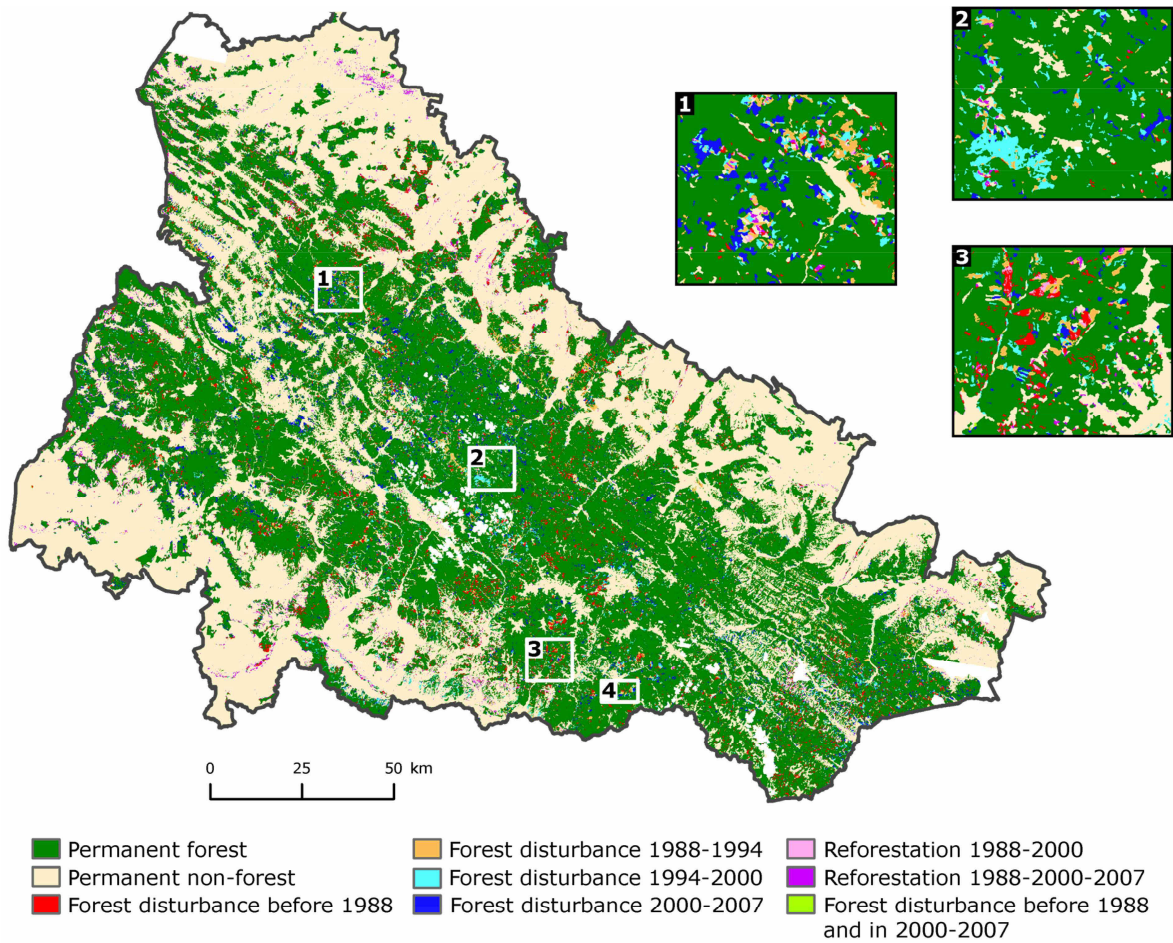


Figure A-2: Top: Forest cover changes between 1988 and 2007. Bottom: Comparison of the satellite-based forest cover change map, stand-level forest management maps, and Landsat TM/ETM+ images from 1994, 2000, 2007 for an area close to Rakhiv, Zakarpatska Oblast. Source: Ukrainian National Forestry University (inventory maps).

forest inventory maps, and disturbances were often substantially larger than reported (Figure A-2, bottom).

Annual disturbance across the study region remained nearly constant until 1994, but dropped markedly (by 39%) in 1994-2000 and increased again after 2000 (by 34%, Figure A-3A). Reforestation rates were four times higher in 2000-2007 compared to 1994-2000 (Figure A-3B). Forest trends differed markedly among the four oblasts (provinces). Disturbance rates in Ivano-Frankivska Oblast increased in 1988-1994 (by 52%), remained stable in Chernivetska Oblast, and decreased in Lvivska and Zakarpatska Oblasts (Figure A-3C). In contrast to the other three provinces, disturbance rates in Zakarpatska Oblast decreased gradually in 1988-2007 (by 32%). Reforestation trends also differed among oblasts. Reforestation rates in Lvivska Oblast and Zakarpatska Oblast were about eight times higher in 2000-2007 compared to 1994-2000, but increased only moderately in Chernivetska Oblast and Ivano-Frankivska Oblast (Figure A-3E). Overall, forest cover increased by 1.29% of the oblast area (equaling 88 km²) in Lvivska Oblast, by 0.83% (23 km²) in Chernivetska Oblast, and by 1.26% (151 km²) in Zakarpatska Oblast, whereas there was a net forest cover decrease of 0.14% (12 km²) in Ivano-Frankivska Oblast.

Disturbance rates also displayed marked heterogeneity at the raion (district) level (Figure A-4). Raions in the interior Carpathians generally exhibited increasing disturbance rates (e.g., Turka and Skole in Lvivska Oblast, Rozhniativ and Bohorodchany in Ivano-Frankivska Oblast, or Putyla in Chernivetska Oblast), but more peripheral raions generally showed decreasing rates (e.g., Drohobych and Stryi in Lvivska Oblast or Tiachiv and Vynohradiv Zakarpatska Oblast). Disturbance rates generally dropped in 1994-2000, but some raions displayed increasing disturbances (e.g. in the East of Zakarpatska Oblast). And the 2000-2007 increase in disturbance rates was most pronounced in the western interior Carpathians (Figure A-4A). High reforestation rates were generally associated with peripheral raions (Figure A-4B), and as a result, peripheral raions dominantly increased forest cover, whereas almost all raions in the interior Carpathians lost forest cover from 1988 to 2007 (Figure A-4C).

Disturbance rates also varied substantially with altitude. Before 1988, the highest disturbance rates occurred at lower elevations (<500 m, Figure A-5A). After 1988 higher disturbance rates occurred at higher elevations, and in 1994-2000, the highest rates were found above 1000 m. The extent of disturbances above 1100 m did not vary substantially until 2000, but dropped by about 50% after new forest legislation became effective (Figure

A-6A). Disturbance rates increased on all slopes in 1988-1994, but there was a clear tendency towards steeper slopes (>30 degrees) in 1994-2000 (Figure A-5B). However, the extent of disturbances on slopes steeper than 20 degrees was similar before and after 2000

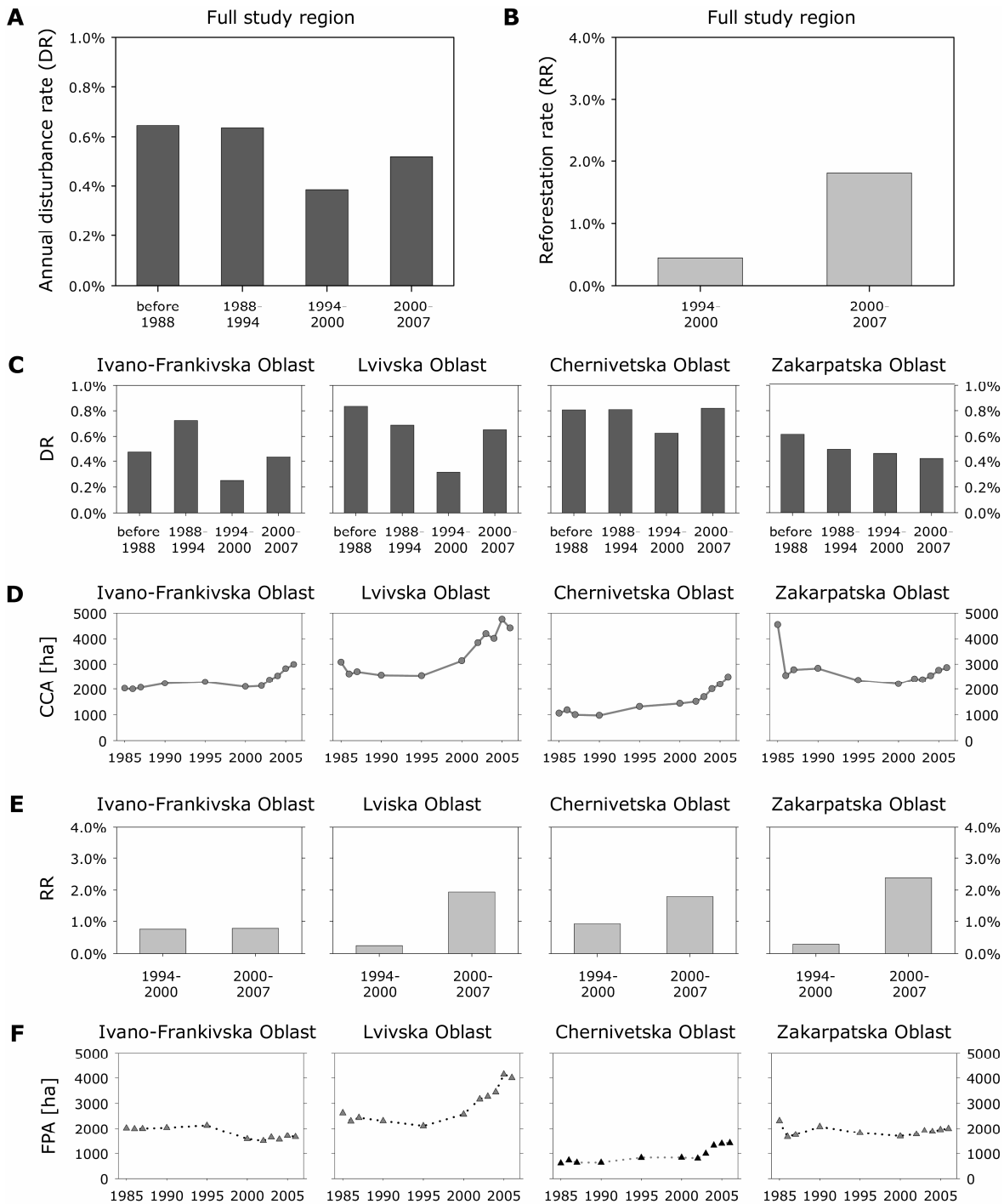


Figure A-3: Remote-sensing-based disturbance and reforestation rates at the study region and oblast level and forest resource statistics at the oblast level from the Statistical Yearbook of Ukraine (2006 and 2007). A: Annual disturbance rates for the full study region. B: Reforestation in the study region (relative to 1988-non-forest land). C: Annual disturbance rates (DR) per oblast. D: Official trends in forest regeneration (i.e., clear-cut) area (CCA) per oblast. E: Reforestation rates (RR, relative to 1988-non-forest land) per oblast. F: Official trends in forest planting area (FPA) per oblast. Source: The State Statistics Committee of Ukraine 2006, 2007 (forest resource statistics).

(Figure A-6B). Last but not least, the proportion of disturbances visible from major roads and railway tracks changed markedly through time. Already before 1988, the majority of disturbances (53%) occurred in invisible areas. After 1988, the proportion of visible forest area increased slightly, yet disturbances increasingly tended to occur in invisible areas. This trend reversed in the time period from 2000-2007 when the proportions of forest and disturbances in visible and invisible areas were approximately equal (Figure A-6C).

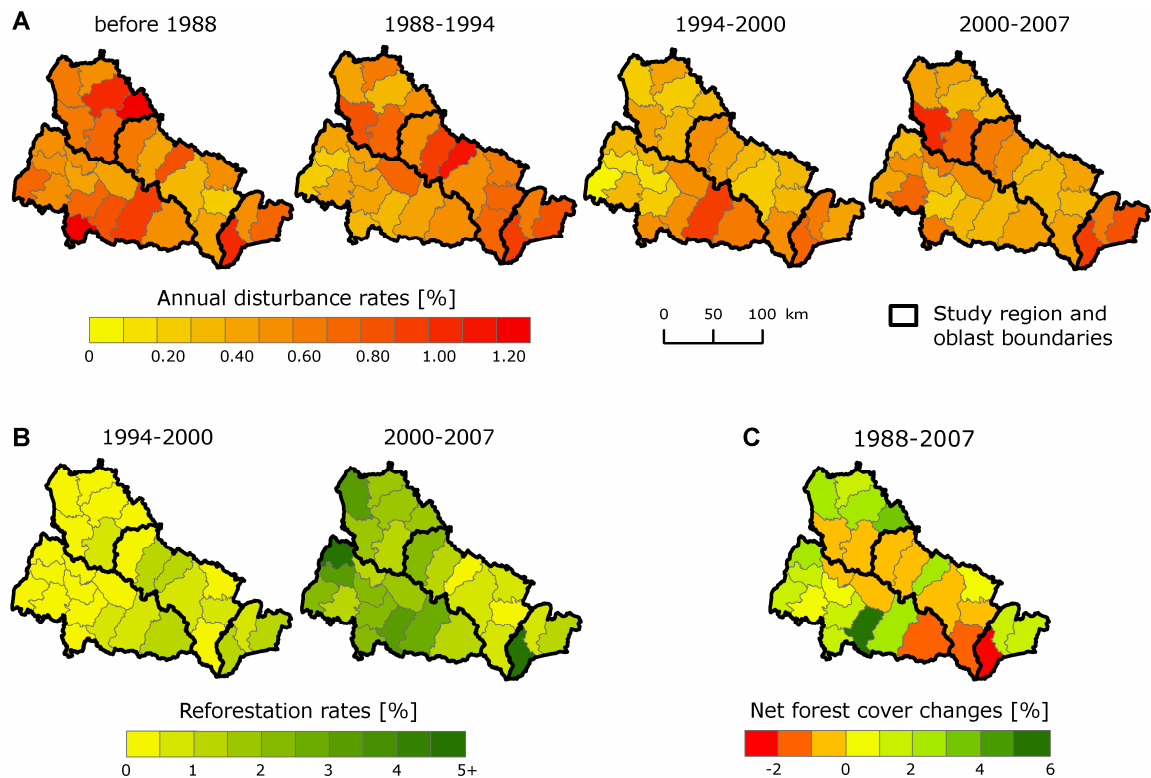


Figure A-4: Disturbance rates (*DR*), reforestation rates (*RR*), and relative net change (*RNC*) rates at the raion level. A: Annual disturbance rates. B: Reforestation in the study region (relative to 1988-non-forest land). C: Net forest cover change (relative to raion area). Source: Geodezkartinformatyka 1997 (oblast and raion boundaries).

Forest cover trends mapped from Landsat images differed markedly from official forest resource data at the oblast level. According to official statistics documented in the Statistical Yearbook of Ukraine, the area of clear-cuts was relatively low until 2000, and increased in most oblasts after 2000. While the post-2000 increase in the statistics was paralleled in our forest disturbance rates in all but Zakarpatska Oblast, the relatively high disturbance rates we found before 1994 in Lvivska and Chernivetska Oblasts and the marked increase in disturbance rates in Ivano-Frankivska Oblast after 1988 were not depicted in the forest resource data (Figure A-3D). Likewise, forest planting trends in forest resource data differed markedly from satellite-based reforestation trends (Figure A-3F).

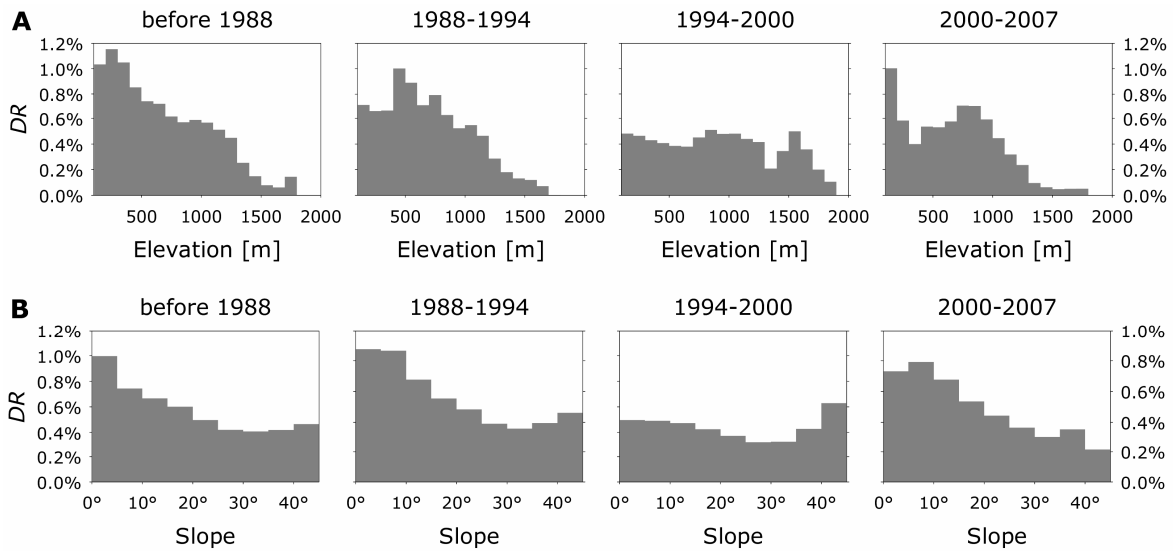


Figure A-5: Changes in disturbance rates by elevation (A) and slope (B).

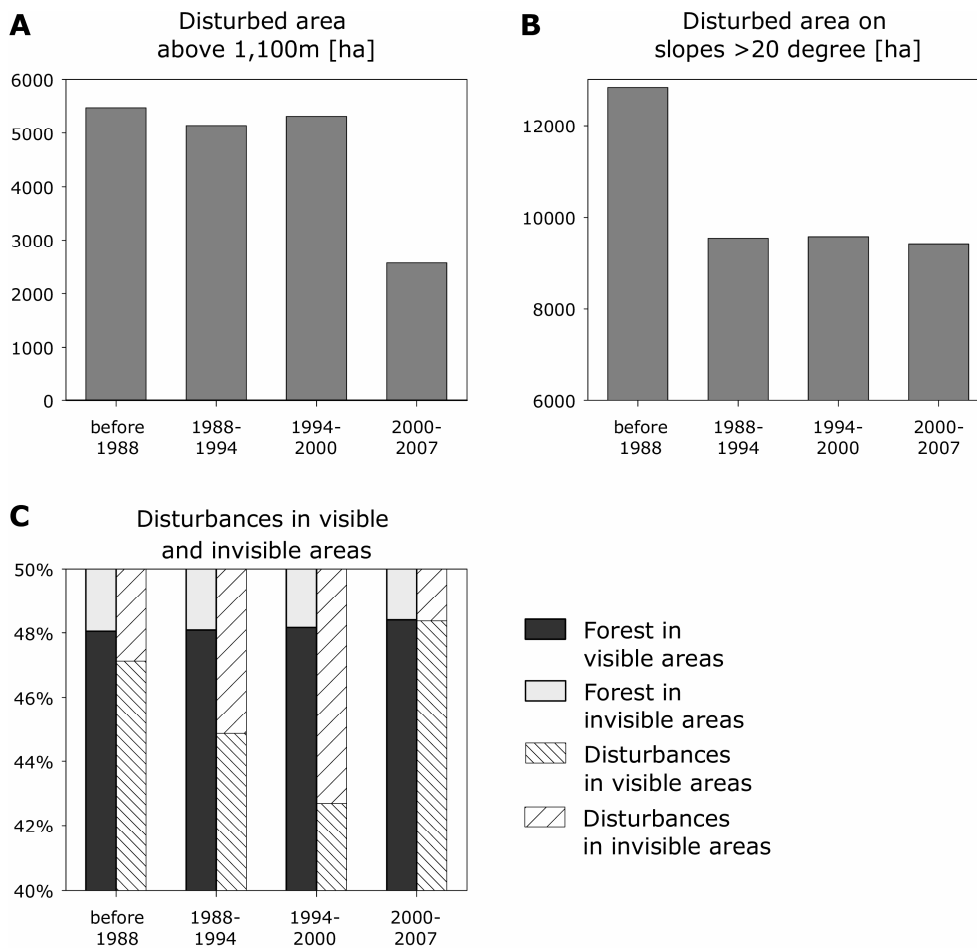


Figure A-6: Disturbed area above and below 1100 m elevation (A), and on slopes >20 degree and <20 degree (B). Proportions of forest and disturbances in areas visible or invisible from major roads and railway tracks (C).

Similarly, the vast majority of forest disturbances before 2000 mapped from the Landsat images were not documented in the inventory map of Zakarpatska Oblast (Table A-4). After 2000, about 34% of all disturbances were detected in areas designated as clear cuts. However, 23% (2318 ha) of all disturbances in 2000-2007 were found in areas where the inventory maps indicated only partial harvesting, and 43% (4302 ha) occurred where officially no forest management had taken place. Also, more than 5698 ha of clear cuts in the inventory maps remained unchanged forest based on the classified Landsat images (Table A-4).

Table A-4: Distribution of permanent forest, disturbances, and reforestation mapped from Landsat TM/ETM+ imagery within different categories of forest management practices as indicated by the inventory map of Zakarpatska Oblast (in ha).

		Forest inventory map				Sum
		No forest management practices	Forest management without forest cover removal	Partial forest cover removal	Complete forest cover removal	
Satellite-based forest change map	Permanent forest	314,388	9592	63,023	5698	392,702
	Disturbance before 1988	10,307	373	4393	74	15,148
	Disturbance in 1988-1994	4609	2492	3259	124	10,484
	Disturbance in 1994-2000	3886	3517	1555	1877	10,836
	Disturbance in 2000-2007	3696	606	2318	3370	9989
	Reforestation	929	51	216	3	1199
	Sum	337,815	16,630	74,764	11,146	

In the inventory map, regular clear cuts and sanitary clear-cuts were almost equally common (Figure A-7A). Sanitary clear-cuts are harvests in response to tree mortality, mainly due to insect disturbance. Selective logging was not very widespread and very few disturbances occurred in such areas. However, sanitary selective logging covered large areas and we found substantial forest disturbances in sanitary selective logging sites. The majority of the areas designated as clear cuts or selective logging sites in the inventory maps were found to represent permanent forest based on the satellite images (Figure A-7A). Visual comparison of clear-cut polygons and Landsat images revealed that forest cover had been completely removed in only 39% of these polygons. Forest cover had only partially been removed in 49% of all polygons, and no disturbance could be visually identified in 12% of all cases (Figure A-7B).

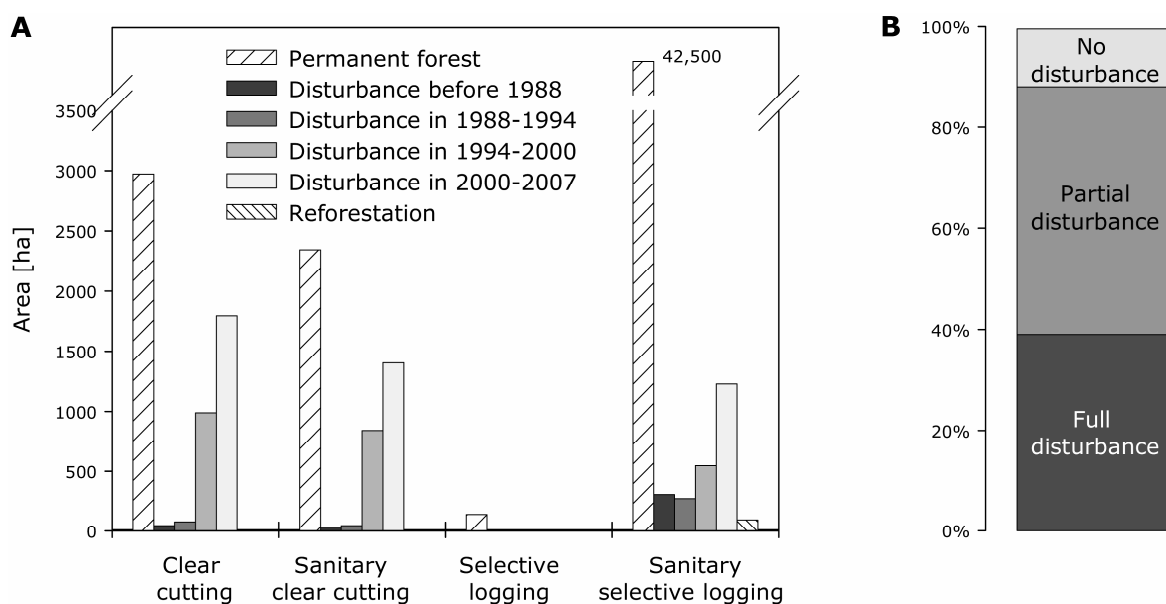


Figure A-7: Distribution of permanent forest, disturbances, and reforestation mapped from the Landsat images for four forest management practices documented in the inventory map (clear cutting, sanitary clear-cutting, selective logging, and sanitary selective logging) (A). Visual assessment of 100 forest management polygons designated as clear-cuts in the inventory data. All polygons were checked against the Landsat images, whether forest cover was intact, partially removed, or fully removed (B).

5 Discussion

5.1 Post-socialist forest cover trends and illegal logging in the Ukrainian Carpathians

Forest disturbance and reforestation resulted in widespread forest cover change in the Ukrainian Carpathians in 1988-2007. The vast majority of disturbances in the study region were due to logging, and forest harvesting trends mapped from satellite images thus differed substantially from forest resource statistics and inventory maps in the Ukrainian Carpathians. What are the reasons for this disagreement? While our accuracy assessments confirmed the high reliability of the Landsat-based change map (see section 5.2 for a detailed discussion of the mapping approach), the inventory data, from which higher-level forest resource statistics are aggregated, exhibited considerable uncertainty. We suggest this uncertainty is the main reason for diverging patterns of satellite-based trends and forest resource statistics.

Updating problems (e.g., where management units were subdivided or merged) and deliberate misreporting cause errors and ambiguity in inventory data (Gerasimov and Karjalainen 2006; Houghton et al. 2007). Even when analyzing only the 2000-2007 period, our results showed that almost 60% of the polygons designated as clear cuts in the

inventory map from Zakarpatska Oblast were only partially harvested or not yet harvested (Figure A-7B), possibly because forest management practices were only applied to a portion of the area of the forest management unit.

Conversely, undocumented logging was widespread in the Ukrainian Carpathians. We found frequent harvesting in areas not designated for harvests as well as over-harvesting beyond the boundaries of designated areas (Figure A-2, Table A-4). While a lack of funding to update inventories may have contributed to these patterns, we suggest illegal logging is the main reason explaining the disagreement between remote sensing and forest inventory maps. After 1991, Ukraine's economy collapsed, state control diminished, and law enforcement, a prime factor in guarding forests from overuse (Chhatre and Agrawal 2008), was weak. Overall, this resulted in emerging shadow business in the Ukrainian forest sector (Nijnik and Van Kooten 2000; Buksha et al. 2003; Nijnik and Van Kooten 2006). Our results indicate that illegal logging may have been especially widespread during the first half of the 1990s, when the discrepancy between satellite-based trends and forest resource statistics was greatest (Figure A-3), and at a time when funds, machines, and fuel were still available to keep forest enterprises running.

Ukraine has since then taken important steps to combat unsustainable forest use. Several protected areas were designated in the Carpathians during the second half of the 1990s, the quality of forest resource statistics improved after 2000, and a new forest code that aims for multi-functional, sustainable forestry, forest certification, and accounting of forest resources, was implemented in 1994 (with important amendments in 2000 and 2006, Nordberg 2007; Soloviy and Cubbage 2007). This clearly affected forest management practices, such as the drop in harvesting above 1100 m and a decrease in disturbances in invisible areas after 2000 (Figure A-6). Moreover, convictions of corrupt forestry staff have recently become public, including the imprisonment of a former head of a forest management enterprise.

Despite these positive trends, corruption continues to be a major problem in Ukraine (Corruption Perceptions Index 2.5/10 in 2007, www.transparency.org). Most importantly, the misuse of the sanitary clear-cut system has emerged as the principal means of illegal logging since the late 1990s (e.g., harvesting of healthy stands, over-harvesting, harvesting in protected areas and at high altitudes, full canopy harvesting in areas designated for selective logging, etc.). Commercial and sanitary logging were almost equally widespread in 2000-2007 in Zakarpatska Oblast. And whereas commercial selective logging frequently

did not show up as disturbance in our forest cover change map, disturbances were widespread in sanitary selective logging sites, likely because forest cover was fully removed in many of these sites. Thus, the sanitary logging system represents a substantial loophole in forest legislation (Contreras-Hermosilla 2002) that is very difficult to monitor. This is exacerbated by the fact that the Ukrainian forest code allows sanitary clear-cuts to be larger than the maximum regular clear-cut size (4 ha). While sanitary logging appears to have been heavily misused, it is important to emphasize that there are also many excellent examples of forest restoration via adequate sanitary logging in the Ukrainian Carpathians.

So what was the extent of illegal logging in the Ukrainian Carpathians after the breakdown of socialism? Uncertainties in the forest inventory data, differences in satellite-based and statistical indicators, and difficulties in separating legal and illegal sanitary logging do not allow answering this question with a hard number. However, four factors suggest that illegal logging may have been at least as extensive as legal logging in the Ukrainian Carpathians. First, forest statistics and satellite-based harvesting rates both showed increased logging after 2000 (when reporting likely improved), but our change map suggests up to 2.8 times higher logging rates before 1994 than documented in the forest resource data (Figure A-3). Second, logging outside areas designated for clear-cuts in the inventory maps was at least as high as in areas declared as clear-cuts. Third, there was still substantial logging above 1100 m, and high-resolution images and field visits suggest that logging in beech-fir forests has not ceased after it was banned. And fourth, substantially higher disturbance rates were observed in areas that were hidden from roads and railways.

Large-scale natural disturbances could offer an alternative explanation for the discrepancy between satellite-based forest trends and forest resource statistics and inventory maps. Wind-throw, root fungi, and insect infestation occur in the Ukrainian Carpathians, but two major factors suggest these processes cannot account for the extent of undocumented disturbances we mapped in the Landsat images. First, most natural disturbances in the Ukrainian Carpathians result only in fine-scale forest cover changes, affecting only single trees or small groups of trees, and our analyses does not map such subtle disturbances (see section 5.2). Large-scale natural disturbances are overall rare (Lavnyy and Lässig 2007; Irland and Kremenetska 2008). For instance, storms represent the region's most frequent natural disturbance, but affected mostly small areas during the last decades and only two extensive wind-throw events (1989 and 1992) were documented (Lavnyy and Lässig 2007). Moreover, most large-scale natural disturbances are associated with spruce plantations that were established during socialism and in Austro-Hungarian times, often on

unfavorable sites (Nilsson and Shvidenko 1999; Badea et al. 2004; Irland and Kremenetska 2008). Higher disturbance rates in such areas are thus at least partly self-inflicted and not a result of natural disturbance regimes (Irland and Kremenetska 2008). Second, where large-scale natural disturbances occur, forest management enterprises almost always carry out salvage logging or sanitary clear-cutting (Lavnyy and Lässig 2007; Irland and Kremenetska 2008), and such disturbances should therefore be documented in the inventory data. Thus, the vast majority of forest disturbance events we mapped from the satellite images were due to forest harvesting, but we can not exclude the possibility that some of these harvests were prompted by natural disturbance events.

Forests were already severely overexploited during socialism, resulting in increasingly younger forests in many areas (Nijnik and Van Kooten 2000; Turnock 2002; Nijnik and Van Kooten 2006). Our results showed a clear tendency towards logging in more remote areas and a net forest cover decrease in the interior Carpathians, likely reflecting an increasing scarcity of high-value timber elsewhere. This raises significant concerns about the fate of Ukraine's Carpathian forest, and especially of ecologically valuable older stands, during the transition. Our results suggest that some regions experienced a net forest cover decrease due to undocumented, illegal logging. This drastically contrasts the popular claim of increasing forest cover in the Ukrainian Carpathians, which recently sparked calls for increased forest harvesting (Polyakov and Sydor 2006).

Reforestation compensated to some extent for high logging rates in the post-socialist period, but mostly in peripheral regions of the Ukrainian Carpathians where much land was managed by state farms prior to 1991. The decreasing profitability of farming frequently resulted in the bankruptcy of these farms, followed by widespread farmland abandonment (DLG 2005). Moreover, Ukraine established a forest planting program in 2002, which may partly explain higher reforestation rates we found between 2000-2007. By and large, however, our results support earlier claims of a slow reforestation in the Carpathians (Kozak et al. 2007a; Kuemmerle et al. 2008; Müller et al. 2008), and only a minor proportion of the region's abandoned farmland has so far reverted back to forests. Reason for this may be that subsistence farming became increasingly important as a livelihood strategy after 1991, particularly in the mountain valleys of the interior Carpathians, and the inconsistent implementation of the national reforestation program.

5.2 Change detection approach

Our change detection approach based on post-classification map comparison of individual forest cover maps yielded a reliable forest change map, which was confirmed by two independent validations (n-fold cross-validation and our disturbance detectability assessment). The n-fold cross-validation we used, widely accepted in other communities (Burman 1989; Burnham and Anderson 1998; Guisan and Zimmermann 2000), has rarely been applied in remote sensing. However, if ground truth is collected via random sampling, n-fold cross-validation results in more robust and conservative error estimates than simply splitting ground truth into a training and validation set (Steele 2005). It is important to note that training and validation data are treated as fully independent datasets each time an error is estimated (i.e., ground truth points used to fit an SVM model are never used to estimate model robustness).

Disturbance detectability was highest in 2000-2007, possibly due to increased logging in spruce plantations after the new forest code was implemented in 2000. Clear cuts in such stands result in higher spectral contrast than in beech/fir forests and are thus easier to map. Although wall-to-wall data did not exist prior to 1988, detection accuracy was similar to 1988-1994 and 1994-2000, suggesting that three post-disturbance images allowed for robust forest regeneration detection. Due to uncertainty in the inventory maps, we digitized disturbance polygons for our validation directly from the Landsat images. While we cannot completely rule out a positive bias, image-based approach typically provide nearly identical results for stand replacement disturbances compared to independent ground truth data (Cohen et al. 1998), and may often be the only option if historic land-cover maps are unavailable. Moreover, traditional ground truth sources (e.g., forest inventory maps, cadastre maps, aerial photos, etc) may be connected to substantial uncertainty, thus introducing a negative bias when assessing the accuracy remote sensing analyses (Foody 2008).

Our results suggest that post-classification map comparisons yield a useful change map if individual classifications are highly accurate and the SVM resulted in very reliable classifications. The non-parametric nature of the SVM allowed us to directly extract thematic classes without having to characterize the substantial spectral variability that existed within these classes due to phenology, illumination, and different land-use systems. Long records of satellite images are becoming increasingly available and our approach may help to move from bi-temporal change detection towards the mapping of trajectories

of change. We suggest post-classification map comparisons may be especially useful in cases where individual classifications are simple (i.e., forest/non-forest), where gathering a representative training set for an integrated multitemporal analyses is not feasible, and where limited data availability precludes full time-series analyses (Kennedy et al. 2007; Röder et al. 2008).

Although our change map was overall highly reliable, a few factors may have contributed uncertainty. Some farmland abandonment may have occurred during socialism (Turnock 2002), which would have inflated pre-1988 logging rates. Likewise, pre-1988 logging rates would be overestimated if forest regeneration took longer than 6 years. However, field visits and prior work (Healey et al. 2005; Kuemmerle et al. 2007) suggest this was not the case, particularly when considering that post-clear-cut planting was carried out prior to the breakdown of the Soviet Union (Buksha et al. 2003). Conversely, we would have underestimated logging rates if regeneration was substantially faster. Field visits render this also unlikely, but we cannot rule out such underestimation completely. It is important to note that underestimation would have affected all time periods similarly and would thus suggest even higher illegal logging rates. Our sampling scheme avoided ground truth points on forest/non-forest boundaries, because positional uncertainty in the Landsat and Quickbird images, and in the non-differential GPS points (<15 m) inhibited us from labeling these points. This could have resulted in overestimated map accuracy, if mixed pixels were widespread in the study region. Yet, the number of discarded points was very low (<3% at most), forest/non-forest boundaries are frequently sharp (even at the timberline) and logging patches are large in the Ukrainian Carpathians, and our validation based on disturbance polygons (which included boundary pixels) confirmed the high accuracy of our maps. Last, our minimum mapping unit of 0.5 ha could have masked fine-scale logging patterns (e.g., fuel wood collection), but was important to remove salt-and-pepper distortions common in pixel-based classifications. While analyzing forest use of local people can give interesting insights (Elbakidze and Angelstam 2007), our focus here was on assessing large-scale forest cover trends (both legal and illegal) which are almost entirely connected to forestry enterprises operating at management units >0.5 ha. Moreover, our minimum mapping unit helped to excluded almost all natural disturbances from our analyses, thus allowing us to separate legal and illegal harvesting.

6 Conclusions

Logging and reforestation on abandoned farmland resulted in widespread forest cover changes in the Ukrainian Carpathians after the breakdown of the Soviet Union. We observed a slight forest cover increase for the entire Ukrainian Carpathians, and the two converse forest change processes led to substantial variability in fine-scale forest cover trends. Peripheral areas, characterized by a high share of pre-1991 farmland, experienced forest cover increase, whereas forest cover decreased in many regions in the interior Carpathians. We also found a clear tendency towards logging in more remote areas and at higher altitudes in the post-socialist period.

Forest trends mapped from Landsat images differed substantially from forest resource statistics and inventory maps. Logging rates did not drop, as suggested by official statistics, during the first years after the breakdown of socialism. To the contrary some regions experienced increased logging. Agreement between satellite-based and statistical indicators was better after 2000, when both sources indicated increasing logging trends. Our analyses also showed that the reliability of inventory maps was mixed.

We suggest that reporting and updating problems as well as illegal logging are the main reasons explaining the mismatch between satellite-based and statistical forest trends. Illegal logging appears to have been especially widespread in the early years after the Ukrainian independence and was likely at least as extensive as legal logging. Ukraine has taken important steps towards sustainable forestry in recent years, and reporting and forest monitoring have improved significantly. Yet, the sanitary clear-cut system remains a major loophole in forest legislation that is almost impossible to control and likely misused for illegal logging (e.g., more timber was logged on sanitary clear-cuts than on commercial clear-cuts in 2000-2007). Overall, our results suggest that unsustainable forest use from socialist times has persisted in the post-socialist period, resulting in continued loss of older forests and their services, and the ongoing fragmentation of some of Europe's last large mountain forests. Transitioning towards sustainable use of these forests and combating illegal logging requires better and up-to-date accounting of forest resources. Remote-sensing-based monitoring can be a key to achieving these goals in the Carpathians and elsewhere in Eastern Europe and the former Soviet Union.

Acknowledgements

We would like to thank A. Rabe and S. van der Linden for the imageSVM implementation and their technical advice. The software imageSVM is available for free at www.hu-geomatics.de. P. Angelstam, M. Elbakidze, M. Dubinin, and O. Krankina provided insightful comments on forest management, forest inventories, and illegal logging in the former Soviet Union. J. Kozak, S. Schmidt and two anonymous reviewers are thanked for very helpful and constructive. We gratefully acknowledge support by the Alexander von Humboldt Foundation, the German Academic Exchange Service (DAAD), and the Land-Cover Land-Use Change (LCLUC) Program of the National Aeronautics and Space Administration (NASA).

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Appendix B:
**Carbon implications of forest restitution in
post-socialist Romania**

Environmental Research Letters 6 (2011)

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Abstract

The collapse of socialism in 1989 has triggered a phase of institutional restructuring in Central and Eastern Europe. Several countries chose to privatize forests or to return them to pre-socialist owners. Here, we assess the implications of forest restitution on the terrestrial carbon balance. New forest owners have strong incentives to immediately clearcut their forests, resulting in increased terrestrial emissions. On the other hand, logging has generally decreased after 1989 and forests are expanding on unused or abandoned farmland, both of which may offset increased logging on restituted forests. We mapped changes in forest cover for the entire country of Romania using Landsat satellite images from 1990 to 2010. Together with historic data on logging rates and changes in forest cover, we use our satellite estimates to parameterize a carbon book-keeping model to estimate the terrestrial carbon flux (above- and below-ground) as a consequence of land-use change and forest harvest. High logging rates during socialism resulted in substantial terrestrial carbon emissions and Romania was a net carbon source until the 1980s. After the collapse of the Soviet Union forest harvest rates decreased dramatically, but since restitution laws were implemented they have increased by 60% (from $15,122 \pm 5397$ ha/y in 2000 to $23,884 \pm 11,510$ ha/y in 2010), but still remained lower than prior to 1989. Romania currently remains a terrestrial carbon sink, offsetting $7.6\% \pm 2.5\%$ of anthropogenic carbon emissions. A further increase in logging could result in net emissions from terrestrial ecosystems during the coming decades. However, forest expansion on degraded land and abandoned farmland offers great potential for carbon sequestration.

1 Introduction

Changes in land-use are an important factor in the global carbon cycle (Houghton and Goodale 2004, Bondeau et al. 2007), yet there are substantial uncertainties regarding the magnitude of carbon fluxes related to land-use (Houghton 2010). While emissions from tropical forest clearing have received much attention (Houghton et al. 2000, DeFries et al. 2002, Archard et al. 2002, Hansen et al. 2008), land-use effects on terrestrial carbon budgets in other regions undergoing rapid land-use change remain uncertain. One such region is Central and Eastern Europe (Henebry 2009, Kuemmerle et al. 2010), where the breakdown of socialism in 1989 triggered fundamental institutional and socio-economic changes and a deep restructuring of the region's forestry and agriculture sectors (Lerman et al. 2004, Rozelle and Swinnen 2004, Tornai et al. 2006).

Forest harvesting generally declined during the 1990s as timber markets collapsed, state support diminished, and institutional changes caused uncertainty (UNECE 2005, Leinonen et al. 2008). Since 2000, harvesting rates have been recovering, sometimes reaching or even exceeding late-socialist rates in some areas. In some regions, illegal logging has also increased in the post-socialist period as a result of rising poverty, institutional decay, and weaker law enforcement (Vandergert and Newell 2003, Henry and Douhovnikoff 2008, Kuemmerle et al. 2009). In the agricultural sector, price liberalization, diminishing markets for agricultural products, declining rural populations, and tenure insecurity have resulted in the abandonment of more than 2 million hectares of farmland (Ioffe et al. 2004, Henebry 2009, Baumann et al. 2011). Reforestation (forest recovery on previously non-forested land such as farmland) on these former farmlands is now common across Eastern Europe and the former Soviet Union (Peterson and Aunap 1998, Leinonen et al. 2008, Kuemmerle et al. 2011).

Although these land-use trends likely altered carbon budgets profoundly, the net terrestrial carbon flux during the post-socialist era remains unclear. The few existing studies have primarily focused on single land-use processes, for example cropland-grassland conversions (Larionova et al. 2003, Vuichard et al. 2008, Vuichard et al. 2009) or logging (Bergen et al. 2003, Krankina et al. 2004). Likewise, most studies assess carbon fluxes in European Russia, while rates of land-use change vary substantially across Eastern Europe (Ioffe et al. 2004, Knorn et al. 2009, Baumann et al. 2011). Finally, existing work has mainly relied on extrapolating field measurements over short time-intervals. Because the

legacies of past land-use can be strong, understanding changes in carbon budgets in the post-socialist period requires reconstructing carbon fluxes over longer time periods (Gimmi et al. 2009, Rhemtulla et al. 2009).

A major problem for assessing carbon fluxes in Eastern Europe is incomplete knowledge about the rates and spatial patterns of post-socialist land-use changes. Forest inventory data from the region are sometimes of low or unknown reliability (Nijnik and Van Kooten 2006, Houghton et al. 2007), and these data often neither account for illegal logging nor reforestation on former farmland. Likewise, estimates of the extent of abandoned farmland vary drastically among different sources (Ioffe et al. 2004, EBRD 2008, FAO 2008). Remote sensing can provide robust assessments of both forest cover change and farmland abandonment (Bergen et al. 2003, Kuemmerle et al. 2008, Kovalskyy and Henebry 2009), but we are only aware of two studies from our own previous work that have used remote sensing to reconstruct carbon dynamics in Eastern Europe. Both studies combined Landsat-based change detection with historic land-use statistics to parameterize a carbon-bookkeeping model, revealing that farmland abandonment resulted in vast carbon sequestration in the Ukrainian Carpathians (Kuemmerle et al. 2010), and that Georgia's forests remain a strong carbon sink despite surging fuelwood use (Olofsson et al. 2010). While these studies highlight the useful insights such approaches can provide, they also emphasize that country-specific policies and institutions strongly affect carbon fluxes. Additional studies focusing on different institutional settings are urgently needed to better understand the carbon dynamics of Eastern Europe in the post-socialist era.

Most importantly, several Eastern European countries chose to return forest to former owners (Sikor 2004, Bouriaud and Schmithuesen 2005b, Ioras and Abrudan 2006, Salka et al. 2006), but the effect of forest restitution on carbon fluxes remains unassessed. Romania is a prime example of a country that chose to reconstitute its forests, (Lawrence and Szabo 2005, Ioras and Abrudan 2006, Lawrence 2009). This process included three phases: the first restitution law (18/1991) returned a total of 350,000 ha (Vasile and Mantescu 2009), the second law (1/2000) targeted another 2 million ha, and the third and final law (247/2005) restituted all remaining forests that were privately owned prior to World War II. Together, 70% of all Romanian forestland has been or will be transferred into non-state ownership, doubling the number of individual forest owners from >400,000 in 2000 (Ioras and Abrudan 2006, Abrudan et al. 2009). Romania's forest restitution process proved complex and the transition period was characterized by substantial economic hardships and tenure insecurity. The incentive of new owners to clearcut their forests is high and

supporting institutions and forest law enforcement are weak. As a result, much concern has been expressed about surging forest exploitation by new forest owners (Bouriaud 2005, Nichiforel and Schanz 2009, Strimbu et al. 2005). On the other hand, forest harvesting rates in state forests were relatively high during socialism and have declined since (Turnock 2002), and much farmland was abandoned in post-socialist Romania (Baur et al. 2006, Kuemmerle et al. 2009b). Both of these effects increase carbon storage in terrestrial ecosystems and potentially counteract anthropogenic carbon emissions from burning fossil fuels. Furthermore, Romania became a member of the European Union in 2007, requiring new forest legislation and management practices, and a substantial enlargement of its protected area network. How institutional changes in the post-socialist period have affected forest cover and thus carbon fluxes remains unclear.

Our aim was to assess the effect of forest restitution in Romania on the terrestrial carbon balance. Our first goal was to map changes in forest cover for the entire country of Romania between 1990 and 2010 using Landsat satellite images. Second, we combined satellite-based estimates of land-use change with historical data on land-use to assess carbon dynamics for the last 200 years using a book-keeping model (Houghton et al. 1983). Our third goal was to assess potential future land-use effects on Romania's terrestrial carbon budget for a range of plausible scenarios of forest harvesting and reforestation.

2 Methodology

2.1 Remote sensing

Forest cover loss was mapped across Romania between 1990-2000 and 2005-2010 using 17 Landsat TM/ETM+ images at a spatial resolution of 28.5 m. The 1990-2000 map was generated using a neural network classifier as described in detail in Olofsson et al. (2010) and Woodcock et al. (2001). The 2005-2010 map was generated by mapping the forest areas of Romania in 2005 and 2010 using a Support Vector Machines classifier and chain classification, and then overlaying these maps to find forest change. A detailed description of this approach is provided in Kuemmerle et al. (2009b) and Knorn et al. (2009). The map categories considered were stable forest, stable non-forest, forest to non-forest, non-forest to forest and other including cloud, cloud shadow and snow. Regrowing forest (non-forest to forest) was excluded from the analysis because of low accuracy and difficulty of

detection using Landsat data. Both maps were subject to rigorous accuracy assessments, based on a stratified random sample of ground reference points independent from the training data (1368 and 1143 samples for the 1990-2000 and 2005-2010 maps, respectively). The samples were interpreted using Google Earth high-resolution imagery in combination with the original Landsat imagery and user's, and producer's accuracy were calculated. Forest change estimates were adjusted according to the error matrix and 95% confidence intervals were calculated for each map category (Cochran 1977, Card 1982).

2.2 Carbon modeling

We employed a well-established carbon book-keeping model to estimate the effect of land-use change on Romania's terrestrial carbon budget. The model tracks changes in carbon stocks (terrestrial and soil carbon) over time as a consequence of three land-use events: (1) deforestation, (2) forest expansion, and (3) logging (and subsequent recovery), each of which is connected to specific release and uptake functions. In addition, parameterizing the book-keeping model requires characterizing the carbon content of mature and disturbed forest systems, specifying growth curves for forest regeneration, and decay functions for different carbon pools. A detailed description of the model is available in Moore et al. (1981); Houghton (1987); Houghton and Hackler (1999); DeFries et al. (2002). Model parameterization is described in Kuemmerle et al. (2011).

The model requires annual rates of three kinds of land-use events. Rates of forest harvest were obtained from the remote sensing maps for 1990-2010. We assumed that the forest loss observed in these maps is due to harvesting (natural disturbances occur, but salvage logging is almost always carried out), which implies that logged forests regenerate. We see little evidence of conversion of logged forests to other land-uses. Forest harvesting rates for 1950-1989 were estimated from forestry statistics (using the area of post-logging forest regeneration as a proxy) (MAPDR 2009, Untaru et al. 2011, Marin and Barbu 2011). The statistical reports (SILV. 1-5) are released annually and contain data on harvested volumes and cutting areas. Harvest rates prior to 1950 were not available and we therefore inferred these rates from forest inventory data (ICAS 1984) for average forest biomass, age structure and average growth rate. (ICAS 1984) contains values for forest inventory parameters and, unlike the statistical reports, is not updated on a regular basis.

Second, we derived historical logging rates that would result in the current age distribution of forests (Romania had an even age class distribution in 1990). The estimated rate in 1950 was exactly the same (60,000 ha/y) as the harvesting rate from the regeneration data,

adding confidence to our approach. Based on this result, we defined a growth curve which allows young forest to grow from 5 to 127 Mg C/ha in the first 80 years, and from 127 to 144 Mg C/ha in the next 100 years. Values for recovery times and carbon contents of disturbed and recovered ecosystems were taken from ICAS (1984).

Rates of deforestation and forest expansion on previously non-forested lands were estimated using data on forest area back to 1800 (MAPDR 1990, MAPDR 1998, Toader and Dumitru 2004, Sofletea and Curtu 2007, Anca 2011). Romania experienced several phases of drastic deforestation, most importantly during the 19th century, when forest areas were reduced by 0.5 million ha; in 1919-1930, when about 1.3 million ha of forest were converted to agricultural land; and after World War II, when about 300,000 ha of forest were cleared. In contrast, forest expansion on abandoned farmland has not been extensive during the 20th century, and has been observed only recently. Together, this allowed us to estimate annual rates of deforestation and forest expansion between 1800 and 2010. While our remote sensing analysis covered all of Romania, the official forestry statistics only referred to land managed by the state (the “forest fund”), which in 2000 consisted of about 6.4 million ha of forest. Our remote sensing estimate of forest area in 2000 was 7.3 million ha, and we therefore rescaled all pre-1990 rates to the entire forest area (i.e., assuming that forest changes on state managed land were also representative outside these areas).

Three different carbon decay pools determine the rate of release of the carbon from logged or cleared forest (Moore et al. 1981). Wood in the first pool is consumed immediately (e.g. firewood), and its carbon released within one year after harvest. The second pool contains short-lived wood products, which decay at a rate of 10% a year (e.g. pulpwood, paper and paperboard). Long-lived wood products, such as furniture and building materials, end up in the third pool and are assumed to decay at a rate of 1% per year. To distribute harvested wood among these pools, we used national forest production statistics from the FAO (FAOSTATS 2011), yielding a distribution of 30.7%, 7.2% and 53.1% among the three pools (the remaining 9% was assumed to end up as slash left on site following harvest).

2.3 Scenarios

To explore the effects of alternative plausible futures, we defined a range of scenarios reflecting a number of different logging and forest expansion rates. A major proportion of Romania's forest has been or will be restituted to former owners. How that affects logging rates is unknown, but incentives for new owners for generating immediate income from restituted forest land are high. Thus, although forest harvesting has decreased in Romania

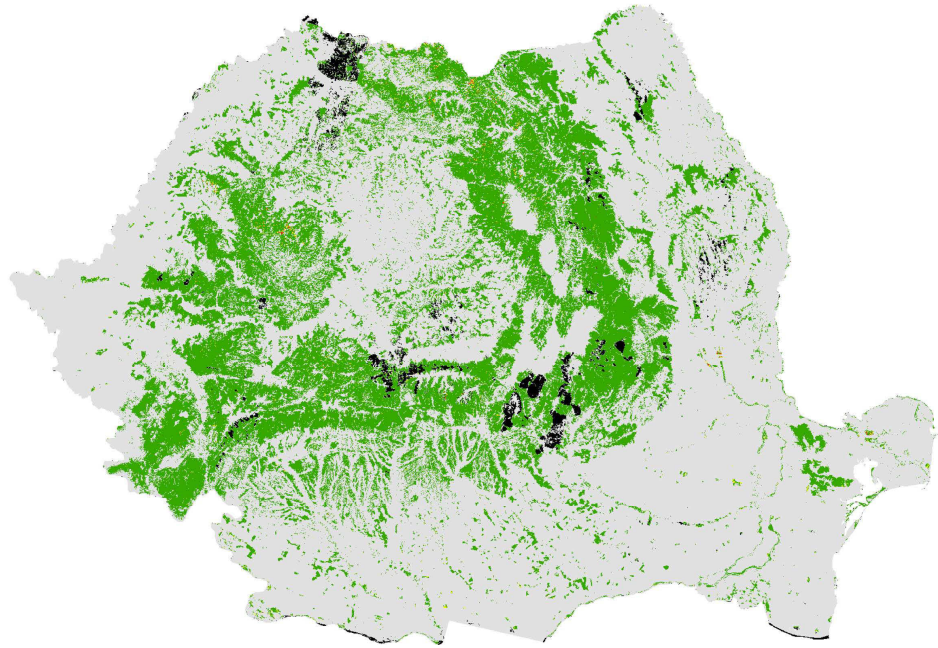
after the collapse of socialism, forest restitution could augment logging rates in the coming decades. Using the current (2005-2010) logging rate as our base rate, we explored five levels of future logging rates for the period 2011-2100: 0% (no logging), 50% (half the current logging rate), 100% (current logging rate), 200%, and 300%. Much farmland in Romania was abandoned after the collapse of socialism or is currently unused. The Romanian Ministry of Agriculture and Rural Development recently reported that 2.9 million ha of farmland is abandoned or currently fallow (MADR 2009). It is unclear how much of this area will eventually revert to forest, but this estimate is extremely important to future carbon fluxes in Romania. Here we include scenarios of 0%, 10%, 20%, 50%, and 75% of forest expansion on the 2.9 million ha of abandoned farmland from 2011 and 2100. The carbon implications of these scenarios were investigated in a full factorial design, resulting in 25 different combinations of logging and forest expansion rates for which the model was run.

3 Results

The annual logging rate estimated from the remote sensing analyses was $15,122 \pm 5397$ ha/y (95% confidence intervals) in 1990-2000 which increased by almost 60% to $23,884 \pm 11,510$ ha/y in 2005-2010. The forest area in 2000 based on our remote sensing analysis was $7,335,448 \pm 379,520$ ha. Assuming that the 2005-2010 logging rate was representative for 2000-2005, the projected forest area in 2010 is 7,096,608 ha, thus resembling closely the forest area in the 2005-2010 map of $6,943,535 \pm 280,693$ ha in 2010. The change map revealed clusters of forest harvesting throughout Romania, especially in Northern Romania (Figure B-1).

The remote sensing analyses yielded reliable forest cover change maps, and the stable forest and non-forest classes were derived with high user's and producer's accuracies in both maps (Tables B-1 and B-2). The logging class had a higher accuracy in the 1990-2000 map (88% user's accuracy) than the 2005-2010 map (51% user's accuracy), likely a result of the post-classification map comparison approach. However, omission errors in the logging class were low for both maps. Because of the relatively small area of logging (<1%), omission errors have a larger impact on the final area estimates, and will result in large confidence intervals. As a result, the confidence intervals for the logging estimates (36% and 48%, respectively) are similar for both change maps.

2005-2010



1990-2000

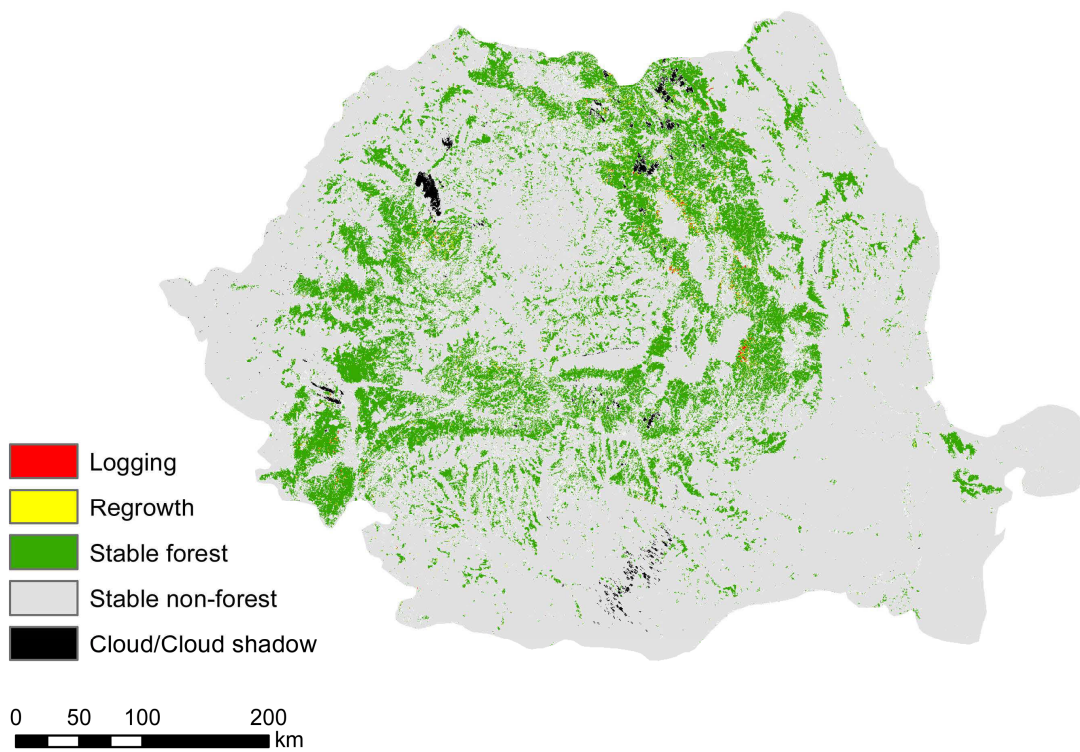


Figure B-1: The two change maps which provided the baseline logging rates between 1990 and 2010. The regrowth class was omitted in the analysis.

Table B-1: The resulting error matrix for the first change map (1990-2000) together with the mapped and adjusted areas and the 95% confidence intervals.

	Logging	Forest	Non-forest	User's acc	N samples
Logging	161	11	10	88%	182
Forest	0	559	4	99%	563
Non-forest	1	49	573	92%	623
Prod's acc	99%	90%	98%		1368

	1990-2000				Annual [ha/y]		
	Map area [ha]	Adj area [ha]	$\pm 95\%$ CI [ha]	$\pm 95\%$ CI [%]	Adj area	Lower CI	Upper CI
Logging	147,290	158,335	56,513	36%	15,122	9725	20,519
Forest	5,995,217	7,335,448	379,520	5%			
Non-forest	17,468,455	16,117,180	382,991	2%			

Table B-2: As Figure B-1 but for the second change map (2005-2010).

	Logging	Forest	Non-forest	User's acc	N samples
Logging	127	66	54	51%	247
Forest	2	322	17	94%	341
Non-forest	0	15	540	97%	555
Prod's acc	98%	80%	88%		1143

	1990-2000				Annual [ha/y]		
	Map area [ha]	Adj area [ha]	$\pm 95\%$ CI [ha]	$\pm 95\%$ CI [%]	Adj area	Lower CI	Upper CI
Logging	154,159	119,420	57,550	48%	23,884	12,374	35,394
Forest	6,846,562	6,943,535	280,693	4%			
Non-forest	16,178,659	16,116,425	275,474	2%			

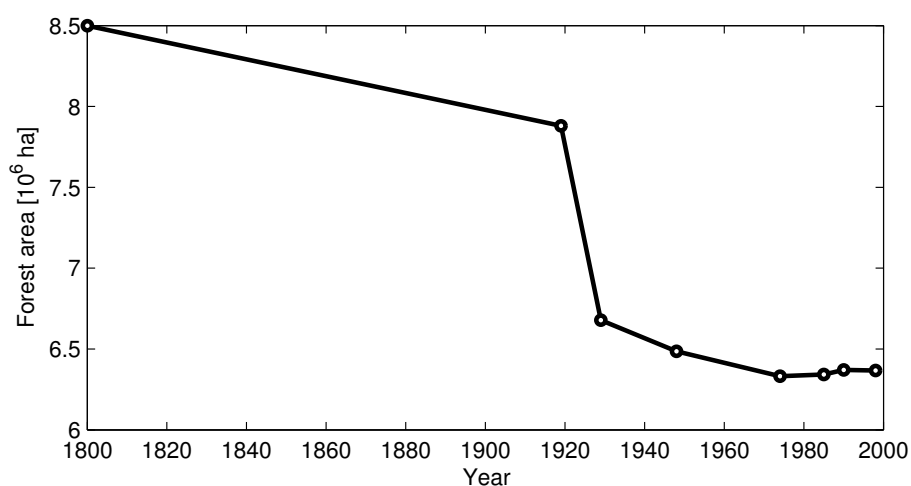


Figure B-2: The forest area of Romania between 1800 and 2000.

Even though we found a significant increase in logging since 2000, current logging rates are substantially lower than logging rates from socialist times. Forest harvesting was

especially widespread during the late 1960s and 1970s, when almost 60,000 ha of forest were logged annually (Figure B-2).

The area of forest in Romania decreased substantially from 1800 until the 1970s (>2 million ha) when the forest cover reached its minimum. The forest cover has remained more or less stable since then (Figure B-2). Forest area decreased throughout the 19th and 20th centuries, except for a small gain just before the collapse of the Soviet Union.

The baseline rates of logging and forest change between 1800 and 2010 are shown in Figure B-3.

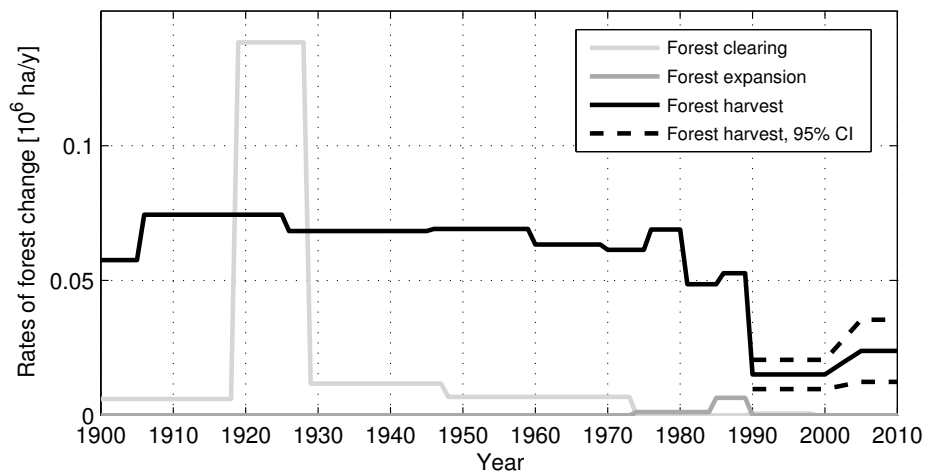


Figure B-3: Baseline input to the carbon book-keeping model. The dashed lines are the 95% confidence intervals for the logging rates estimated from satellite data.

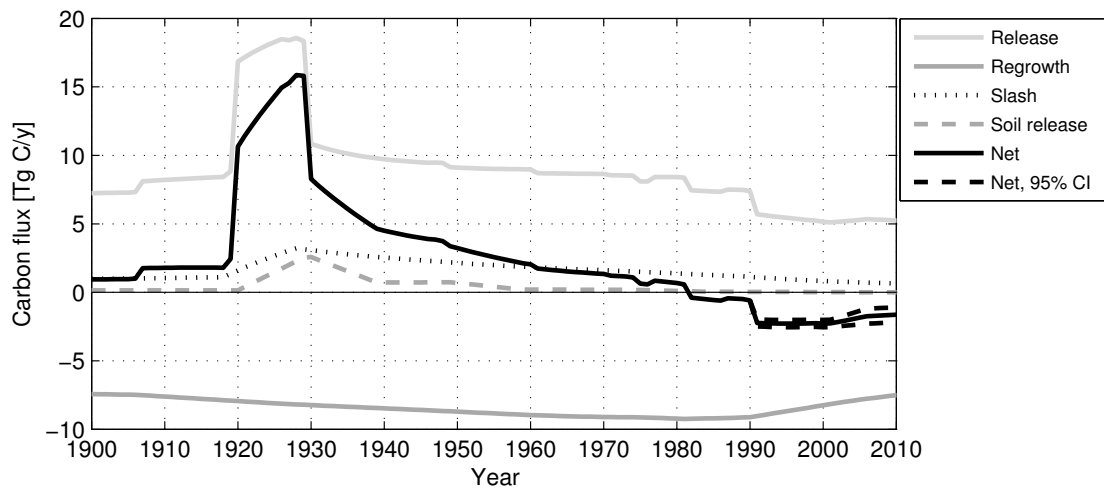


Figure B-4: Terrestrial carbon flux in Romania as a result of the baseline rates in Figure B-2. As the model only associates release and uptake of soil carbon with permanent forest loss and gain, the soil carbon flux is close to zero and therefore not plotted. (A positive flux equals terrestrial emissions.)

Table B-3: The offset in 2050 for the 25 different scenarios using the current anthropogenic carbon emissions.

Logging	Forest Expansion				
	0%	10%	20%	50%	75%
300%	–	–	–	–	3%
200%	–	–	–	3%	6%
obs	–	1%	3%	7%	10%
50%	2%	3%	4%	8%	11%
0%	4%	5%	6%	10%	13%

Reconstructing carbon fluxes due to land-use change and logging revealed that Romania has been a net carbon source throughout much of the 20th century (Figure B-4). Terrestrial emissions were highest in 1920-1930, as a result of the massive deforestation at that time. During socialism, terrestrial emissions gradually declined despite relatively high logging rates - mainly because of carbon sequestration in regenerating forests. The terrestrial carbon balance shifted from a source to a sink in the 1980s, and has remained a net sink throughout the post-socialist period. However, increased logging in the post-socialist period is reflected in the carbon flux by diminished strength of the sink after 2000 (Figure B-4). Currently (2010), Romania's net terrestrial carbon sink is 1.64 Tg C/y (Figure B-4), an 7.6% offset of Romania's anthropogenic carbon emissions (US Energy Information Administration 2011). The lower and upper confidence intervals of the logging estimates generate sinks in 2010 ranging from 1.10 to 2.18 Tg C/y, which corresponds to anthropogenic emissions offsets of 5% and 10%. Thus, we estimate the current terrestrial carbon sink to be 1.64 ± 0.54 Tg/y which equals an offset of $7.6\% \pm 2.5\%$.

Alternative scenarios of future logging rates and forest expansion rates on currently unused lands show significant effects on net flux of carbon for the period 2011-2100 (Figure B-5). Romania remained a carbon sink throughout the 21st century for many of the 25 scenarios we assessed, especially if logging rates remain at current levels or lower (Figure B-5). In contrast, higher logging rates would shift Romania from a sink to a source within the next decades. Forest expansion on currently unused land (either by way of natural succession or afforestation) could substantially offset higher terrestrial emissions from logging. For example, assuming a logging rate twice the current rate (Figure B-5b), could either result in net carbon emissions (e.g., if only 10% of all unused land reverts to forest, second lightest grey line) or sequestration (e.g., if forests regrow on 50% of all currently unused land, second darkest grey line in Figure B-5b). Assuming no logging and no forest expansion throughout the 21st century would result in a source of 0.76 Tg C/y in 2100 but a total sink of 56 Tg C between 2011-2100 (Figure B-5e). In contrast, a threefold increase

of the 2005-2010 logging rates after 2011 in combination with no forest expansion would result in a net carbon sink of up to 0.019 Tg C/y in 2100 but a total source of 84 Tg C between 2011-2100 (Figure B-5a). Assuming that the observed rates of logging and forest change remain constant throughout the 21st century would result in a sink until about 2050 after which it would turn to a source, and a total sink of 9 Tg C for the remainder of the century (Figure B-5c). Depending on the harvesting and forest expansion rate, Romania's forests could compensate for up to 13% of the country's anthropogenic carbon emissions (Table B-3).

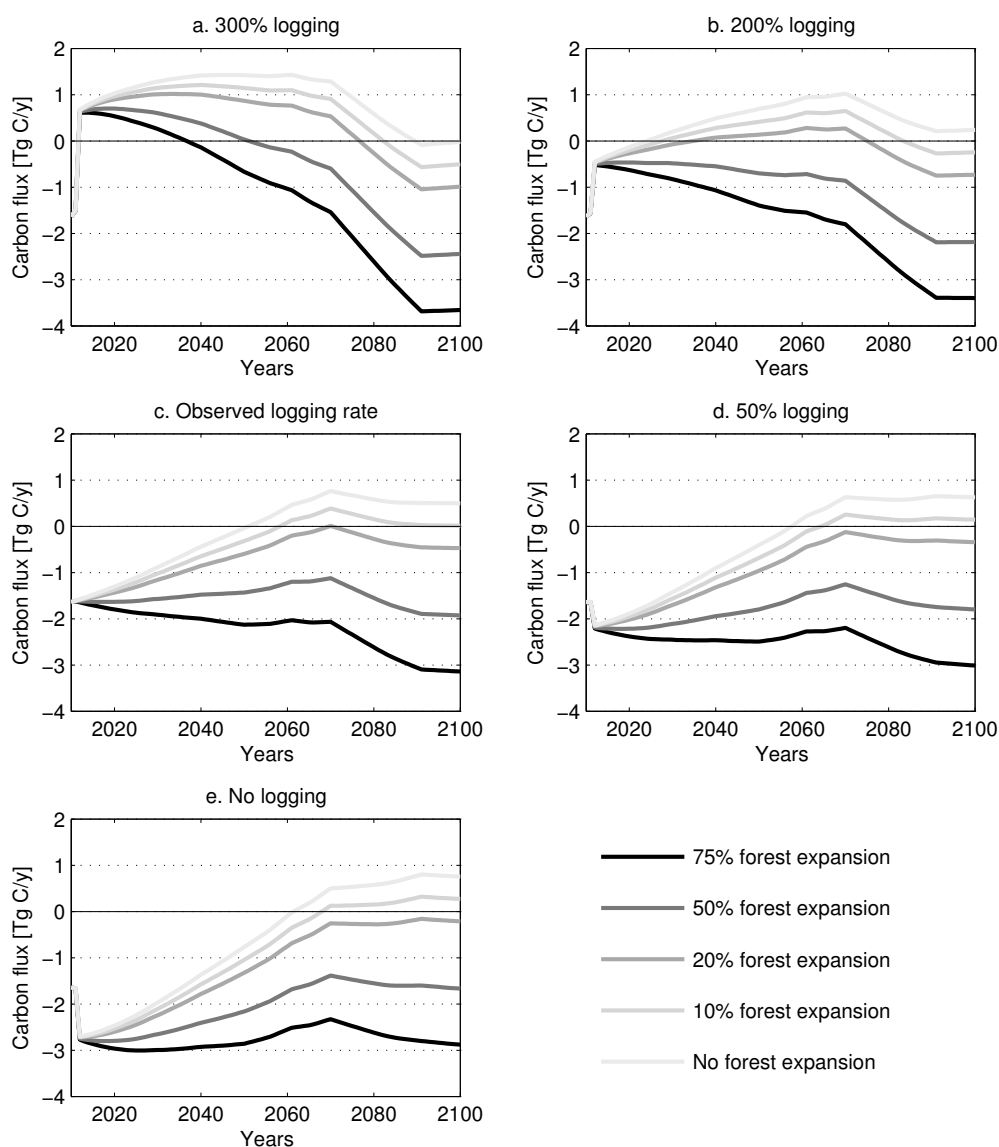


Figure B-5: The net terrestrial carbon flux when running the model with 25 combinations of different logging and forest expansion rates. Each of the five plots represents a logging scenario. Figure B-5a (“300% logging”) shows the terrestrial carbon flux for a threefold increase of the current logging rate between 2011 and 2100, while Figure B-5c (“Observed logging rate”) shows the flux if the current rate is kept constant until 2100. The lines in each plot represent different rates of forest expansion on non-forested lands. Increased logging results in higher initial release but also higher sequestration at the end of the century. Higher rates of forest expansion rates result in dramatically increased carbon sequestration.

4 Discussion

The collapse of socialism profoundly affected Romania's land-use systems, and in turn, the terrestrial carbon budget. Romania implemented one of the most dramatic forest restitution policies across Central and Eastern Europe, transferring up to 70% of all forests from state into private ownership. Our results suggest that these ownership transfers resulted in a 60% increase in logging rates, and a measurable effect on the countries net carbon flux from land-use. Our study thus provides further support to previous studies that report that changes in forest property rights can trigger excessive resource use (Mena et al. 2006, Deacon 1999, Strimbu et al. 2005). Three factors explain increased logging rates after forest restitution laws were implemented. First, the transition period was characterized by substantial economic hardships (e.g. Romania's GDP has not recovered to pre-1989 levels) providing an incentive to many new owners to immediately clearcut their forests for short-term returns (Ioras and Abrudan 2006, Strimbu et al. 2005). Second, Romania's forest restitution was a slow and complex process, with many new owners fearing that their property rights were not permanent (Ioras and Abrudan 2006, Sikor et al. 2009). Third, the post-socialist period in Romania was, as elsewhere in Eastern Europe, characterized by decreasing transparency, lower institutional strength, and weak law enforcement, resulting in increasing illegal logging (e.g., timber theft) and a lack of conformity with forest laws (e.g., over-harvesting, harvesting inside protected areas) (Ioja et al. 2010, Ioras and Abrudan 2006, Strimbu et al. 2005, Turnock 2002, Irland 2008).

The carbon implications of restitution of forest to pre-World War II owners was small compared to terrestrial emissions resulting from logging during the socialist period. Despite increased forest harvesting rates in 2005-2010 compared to 1990-2000, logging rates are considerably lower than those prior to 1989. Socialist-era logging was particularly intensive in the 1960s and 1970s. During that time, maximum utilization of natural resources was the main land-use paradigm in many socialist countries, often leading to unsustainable resource use (Turnock 2002) (in the case of forest due to massive development of the woodworking industry). Although excessive logging resulted in high initial carbon emissions, regenerating forests on former logging sites also sequestered considerable amounts of carbon. As logging rates gradually decreased during the last years of socialism, Romania shifted from a terrestrial net carbon source to a net carbon sink. Considering the relatively long time of sustained growth (180 years) of regenerating forests on former logging sites, Romanian forests will continue to sequester carbon throughout the first half of the 21st century. This result highlights the long-lasting legacy of socialist-era

forest management on today's carbon budgets (Main-Knorn et al. 2009, Kuemmerle et al. 2010).

Forest harvest in Romania dropped markedly after 1989 (Figure B-3) and this further accentuated the ongoing carbon sink. As elsewhere in Eastern Europe (Bergen et al. 2008, Kuemmerle et al. 2007), timber markets collapsed and prices for inputs and outputs were liberalized. In addition, the early post-socialist years were characterized by substantial tenure insecurity and harvesting of forests designated for restitution was sometimes stopped (Abrudan and Parnuta 2006). The immediate effect of decreasing forest harvests on the terrestrial carbon budget is mainly defined by the forest growth curve and the allocation of wood products to the carbon decay pools. In Romania's case, 31% of the carbon is released immediately, which explains the drop in terrestrial emissions after 1989. Lower harvesting rates since 1989 resulted both in foregone emissions and an increasing growing stock for Romania's forests, both increasing the magnitude of the carbon sink during the post-socialist period.

The current sink strength is furthermore notable considering that the extent of forestland remained nearly constant, both during socialism and in the post-socialist period. This is remarkable as much farmland was abandoned or set aside after the collapse of socialism. Quantitative data on the extent of farmland abandonment is scarce, but almost 3 million ha of farmland have been reported being out of production as of 2009. Several reasons explain why only a small proportion of these lands have reverted to forest since 1989. First, much of these former farmlands may not be permanently abandoned or are still being used for occasional grazing. Second, abandoned farmland may be degraded (up to 400,000 ha of such degraded lands exist throughout Romania) (Abrudan et al. 2009), thereby inhibiting spontaneous forest regeneration. Third, abandoned or set-aside land may occur far away from existing forests (e.g., in Romania's plains, where forest cover has been dramatically decreased historically), thus retarding succession to woody communities. Last, afforestation rates were very low in Romania until a systematic afforestation program was initiated in 2005 (Dutca 2011).

The steady decrease of forest cover before World War II and the relative stability thereafter (Figure B-2) also suggest that Romania has not experienced a forest transition, in contrast to many neighboring countries (Kuemmerle et al. 2011, Kozak et al. 2007, Mather 2001). Forest transition theory describes the reversal of deforestation associated with industrialization and urbanization (Mather 1992). In Romania, forest cover appears to have

declined in several phases, most markedly after 1918 when about 1.3 million ha of forest land were given to World War I soldiers with the obligation to farm these lands (triggering a staggering carbon release of up to 19 Tg C during the 1920s, Figure B-4). Although speculative, one interpretation of the missing forest transition pattern is that much of the currently unused farmlands will eventually return to forests, especially those areas that are marginal for farming.

Comparing different scenarios of future logging and forest expansion highlights the variety of plausible carbon flux futures. Even with zero future logging, the current sink strength is projected to decrease over the 21st century. The reason is that carbon storage due to regenerating forests on areas logged during socialism will decrease, while more than half of the wood harvested during that time is still oxidizing because it became long-lived wood products (53% of all wood). With the restitution process not fully finished and forest institutions still in transition, either drastically increasing or declining logging rates are plausible. Our scenarios reveal that even under current logging rates, Romania's carbon sink will only last until about 2050 after which it will convert into a small source for the rest of the century. This shift would occur substantially earlier if logging rates would rise further (e.g. in 2025 for twice the current logging rate, Figure B-5b).

The regrowth of forest on former farmland could help offset terrestrial emissions from logging and maintain or even strengthen the current sink throughout the 21st century even if logging rates increase (Figure B-5). How much of the currently unused farmland will be returned into production remains highly uncertain. Our scenarios highlight the vast carbon sequestration potential on these abandoned farmland, similar to other post-socialist countries (Vuichard et al. 2008, Kuemmerle et al. 2011). Because spontaneous forest development is unlikely for many degraded areas, reforestation and afforestation of abandoned farmland could be an attractive land-use in light of incentives provided by carbon markets (Kuemmerle et al. 2010). Romania has recently established the ambitious goal of expanding forestland in the coming decades by about 2 million hectares. This is close to a forest expansion scenario of 75% of all currently abandoned farmland, suggesting that such a policy would result in a large carbon sink of about 3 Tg/y by the end of this century (even when further increasing forest harvesting). The population of Romania has been decreasing steadily since 1990, with an annual decrease of 90,000 people on average (World Bank 2010). If this trend continues, it is likely to result in increasing rates of farmland abandonment.

A sensitivity analysis of the carbon book-keeping model aiming at investigating the effect of errors in the estimated model parameters was not performed. However, many of the values of the model parameters in this study were also used in Kuemmerle et al. (2011) in which a rigorous sensitivity analysis was performed. Further sensitivity analysis was performed by Houghton (2005).

5 Conclusions

Our study revealed a significant effect of forest restitution on Romania's terrestrial carbon budget and emphasized the significant legacy of socialist land-use and forest harvesting on today's carbon budget. The current carbon sink in the terrestrial ecosystems of Romania is a substantial fraction of the country's anthropogenic emissions, and future forest expansion could substantially increase the current sink strength even under increased logging scenarios. Romania harbors some of Europe's last relatively undisturbed forest ecosystems, and substantial concerns have been expressed about unsustainable forest use triggered by the forest restitution process (Ioras et al. 2009, Ioja et al. 2010, Knorn et al. 2011). While the carbon effects of logging were comparatively small in our study, we urge policy makers and land-use planners to fully account for the trade-offs and synergies between economic returns from forestry, provision of ecosystem services (e.g., flood retention, soil stability), and biodiversity conservation.

Acknowledgements

This research was funded by NASA Award NNG05GG14G and the Belgian Science Policy under the Stereo II research program (contract SR/00/133). TK gratefully acknowledges support by the Alexander von Humboldt Foundation.

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Eidesstattliche Erklärung

Hiermit erkläre ich, die vorliegende Dissertation selbstständig und ohne Verwendung unerlaubter Hilfe angefertigt zu haben. Die aus fremden Quellen direkt oder indirekt übernommenen Inhalte sind als solche kenntlich gemacht. Die Dissertation wird erstmalig und nur an der Humboldt-Universität zu Berlin eingereicht. Weiterhin erkläre ich, nicht bereits einen Dokortitel im Fach Geographie zu besitzen. Die dem Verfahren zu Grunde liegende Promotionsordnung ist mir bekannt.

Jan Knorn

Berlin, den 08. Februar 2012