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Landsat-based mapping of post-Soviet land-use change to assess the effectiveness of the Oksky and Mordovsky protected areas in European Russia

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A B S T R A C T

Land-use and land-cover change (LULCC) is the main cause of the global biodiversity crisis and protected areas are critical to prevent habitat loss. Rapid changes in institutional and socio-economic conditions, such as the collapse of the former Soviet Union in 1991, often trigger widespread LULCC. Yet, it is unclear how effective protected areas are in safeguarding habitat within them during such periods of rapid LULCC. Our goal here was to map changes in forest cover and agricultural lands from 1984 to 2010 in order to assess the effectiveness of two strictly protected areas, Oksky and Mordovsky State Nature Reserves, in temperate European Russia. We analyzed dense time series of Landsat images for three Landsat footprints and applied a support vector machine classification and trajectory-based change detection to map forest disturbance. We then used matching statistics to quantify the effectiveness of the protected areas. Our analyses highlighted considerable post-Soviet LULCC in European Russia. The LULCC maps revealed disturbances on 5.02% of the total forest area, with strongly declining disturbance rates in post-Soviet times. We also found that 39.89% of the agricultural land used in 1988 was abandoned after 1991, leading to widespread forest regrowth. Oksky and Mordovsky State Nature Reserves had a significantly lower probability of forest disturbance (−0.1 to −3.5% lower) in comparison to their surrounding areas. This suggests that protected areas were relatively effective in limiting human-induced forest disturbance in European Russia, despite lower levels of control and an eroding infrastructure for nature protection. Moreover, we found drastic land-cover changes, particularly forest regrowth, in the surroundings of these protected areas, highlighting conservation opportunities. Protected areas can play a key role in biodiversity conservation during periods of rapid LULCC, and remote sensing coupled with matching statistics provide important tools for monitoring the success and failure of conservation efforts.

1. Introduction

Global biodiversity is declining rapidly (Butchart et al., 2010), with land-use and land-cover change (LULCC) and overexploitation being two of the main drivers of these losses (EEA, 2007; Gonzalez et al., 2011; Millennium Ecosystem Assessment, 2005). LULCC affects biodiversity via habitat loss, degradation, and fragmentation (Lindenmayer and Fischer, 2006), and as such represents a challenge for biodiversity conservation in areas where land use is intensifying (Fischer et al., 2012; Kleijn et al., 2009; Rudel et al., 2009). However, land-use change can also result in ecosystem recovery, for example, where shifting socio-economic conditions trigger agricultural abandonment (Benayas et al., 2009; Kuemmerle et al., 2008; Meyfroidt and Lambin, 2011). Both processes can co-occur, leading to complex outcomes. This is particularly the case for regions where socio-economic shocks (e.g., revolutions, wars, or epidemics) take place, frequently leading to both illegal resource use (Greenpeace, 2008; Kuemmerle et al., 2009), and the abandonment of agriculture (Hostert et al., 2011; Pongratz et al., 2011; Yeloff and van Geel, 2007). Better understanding the complex interrelations of socio-economic shocks and LULCC is therefore important to identify efficient biodiversity conservation strategies.

Protected areas are a cornerstone of global conservation efforts (Dudley et al., 2010; Margules and Pressey, 2000; Rodrigues et al., 2004). Many protected areas are both directly and indirectly affected by human land use, either because they permit at least some human use within their territory (Radeloff et al., 2010), or because they are surrounded by intensive land use (Curran et al., 2004). A particular protected area is embedded within a larger ecosystem via a 'zone of

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interaction' (DeFries et al., 2010), highlighting that there are both strong ecological and socio-economic interactions between protected areas and their surroundings (Hansen and DeFries, 2007). These interrelations raise the question how socio-economic shocks, which may erode the infrastructure for conservation (Henry and Douhovnikoff, 2008) and which are known to lead to drastic LULCC, affect protected areas.

One of the most dramatic socio-economic shocks in recent times, in terms of area affected, was the collapse of the Soviet Union in 1991. The subsequent transition from a socialistic-planning to market-oriented economic systems strongly affected forestry and agricultural sectors in almost all succession states of the Soviet Union (Krankina and Dixon, 1992), and this triggered drastic land-use changes. In Russia, the largest country of the former Soviet Union, forest harvesting changed considerably (Torniainen et al., 2006), with decreasing logging rates in some areas, for example, European Russia (Baumann et al., 2012; Potapov et al., 2011; Wendland et al., 2011) and southern central Siberia (Bergen et al., 2008), but also increased illegal logging, for example, in the Russian Far East and eastern Siberia (Vandergert and Newell, 2003). Overall though, logging patterns in post-Soviet Russia are not well understood (Houghton et al., 2007). In terms of agriculture, the dominant trend of land-use change was the widespread abandonment of farmland throughout Eastern Europe (Ioffe et al., 2004; Kuemmerle et al., 2011; Peterson and Aunap, 1998; Prishchepov et al., 2013). Reforestation on abandoned farmland may have increased the total forest area in European Russia (Baumann et al., 2012), but where and how much abandonment and reforestation happened remains also unclear.

Russia is also a particularly interesting country to investigate the effects of LULCC on protected areas because Russia harbors exceptional biodiversity (Pavlov, 2001) and has a well-established and extensive network of protected areas. Today, there are more than 11,000 protected areas covering about 200 million ha, equalling about eleven percent of the Russian territory (IUCN UNEP-WCMC, 2011; Krever et al., 2009). Of these protected areas, 102 are zapovedniks, (i.e., strictly protected, scientific state nature reserves, IUCN category Ia) (IUCN UNEP-WCMC, 2011), established solely for conservation and scientific monitoring. Zapovedniks, particularly the older ones (i.e., the first zapovednik was founded in 1892, Danilina, 2001), preserve unique landscapes across different ecoregions in Russia. More zapovedniks are located in European Russia, but these are usually smaller in size due to the higher human population densities and the long history of intensive land use (Spetich et al., 2009). The collapse of the Soviet Union in 1991 resulted in severe funding cutbacks for conservation efforts, and many protected areas are today short in personnel, equipment, and financial capacity (Wells and Williams, 1998). At the same time, weak law enforcement resulted in increasing illegal resource use in the post-Soviet period, for example, illegal logging (Eikeland et al., 2004; Morozov, 2000) and poaching (Milner-Gulland et al., 2003), thus posing new challenges for conservation. Given these challenges, it remains unclear whether Russia's protected areas remained effective in the post-Soviet period. Likewise, we currently lack knowledge on how LULCC affected protected areas and their surroundings in Russia.

Assessing the spatial patterns of post-Soviet LULCC is challenging though because data on changes in forest cover, such as forest inventory data, are often not easy to access, out of date, unreliable, available only in aggregated form, or lack information on illegal logging (Filer and Hanousek, 2002; Houghton et al., 2007). Similarly, data on changes in agricultural land use are often not available for larger areas and do not provide information on potential forest succession. Remote sensing has therefore become a key technology for monitoring post-Soviet LULCC (Bergen et al., 2008; Kovalskyy and Henebry, 2009; Kuemmerle et al., 2011; Peterson and Aunap, 1998). In the past, most LULCC approaches were limited by the availability and the cost of data, and mostly focussed on bi-temporal change detection. With the recent opening of the USGS Landsat archives, dense time series of satellite imagery are now available for many regions in the world, spanning 30 years of land-use change including the entire post-Soviet period. Newly developed approaches of time series analyses allow assessing changes in a pixel's spectral–temporal profile or of proxies derived from the original spectral data (Huang et al., 2010; Kennedy et al., 2010) to better identify both rapid and gradual LULCC. This provides new opportunities to better understand the effects of socio-economic shocks, which happen at distinct points in time on land systems and on the effectiveness of conservation (Griffiths et al., 2012).

Likewise, remote sensing has been instrumental to measure the effectiveness of protected areas (Curran et al., 2004; Gorsevski et al., 2012; Knorn et al., 2012; Kuemmerle et al., 2007). Such assessments have traditionally often relied on comparing LULCC inside and outside protected areas. This is problematic considering that protected areas are regularly established in marginal or remote areas and that the protection may lead to spillover effects, for example, increased land use pressure in the surrounding areas (Andam et al., 2008). Simply comparing rates of LULCC inside and outside of protected areas may therefore produce incomplete estimates of a protected area's effectiveness if location bias remains unaccounted for (Joppa and Pfaff, 2009). Novel statistical approaches based on matching statistics reduce bias by identifying and comparing pairs of observation points inside and outside the protected area that are most similar to each other based on a list of covariates (Andam et al., 2010). To our knowledge, however, no study has so far combined remote sensing based assessments of post-Soviet LULCC and matching statistics analyses to assess protected area effectiveness anywhere in the former Soviet Union.

There are very few places in the world where LULCC following a socio-economic and institutional shock has been as widespread and rapid as in Russia. While Russia has an extensive and long-established protected area network, the collapse of the Soviet Union gives rise to substantial concerns about the effectiveness of this network (Brandt, 1992). Here, our goal was to quantify post-Soviet LULCC and to assess the effectiveness of two long-established strictly protected areas (zapovedniks) in a region representative for those with LULCC in European Russia: Oksky State Nature Reserve and Mordovsky State Nature Reserve. We analyzed a time series of Landsat images covering the time period between 1984 and 2010 across three Landsat footprints to quantify forest change and farmland abandonment in the post-Soviet period. As a measure of protected area effectiveness, we compared forest disturbance rates inside and outside the protected areas based on matching statistics. Specifically, we had three objectives:

1. to assess the rates and spatial patterns of forest disturbances and subsequent reforestation within and outside the protected areas;
2. to assess the rates and spatial patterns of farmland abandonment and subsequent reforestation; and
3. to evaluate the reserves' effectiveness in preventing loss of forest habitats due to logging and how this relates to the reserves' surrounding land use in European Russia.

2. Methods

2.1. Study area

Our study area was located in European Russia and covered more than 67,000 km^2 within Ryazan Oblast and Mordovia Republic, about 200 km Southeast of Moscow (Fig. 1). Altitude varies from about 100 to 300 m. The climate is temperate-continental, with warm summers (mean July 19.8 °C) and cold winters (mean February −11.6 °C), and mean annual precipitation of 534 mm (Priklonsky and Tichomirov, 1989). The region is part of the temperate broadleaf and mixed forest biome and located at the junction of two ecoregions: the sarmatic mixed forest with boreal forests dominated by spruce (*Picea abies*) and Scots pine (*Pinus sylvestris*) as well as mixed temperate (with oak, *Quercus robur*) forests in the North, and the East European forest steppe with a mosaic of deciduous forests of lime (*Tilia cordata*) and oak (*Q. robur*) areas and steppe vegetation in the South (Olson et al., 2001).

Ryazan Oblast and the Republic of Mordovia are characterized by low population densities (29.1 and 31.6 persons per km^2 in 2010, respectively, Heaney, 2011) and decreasing population size, with a net loss of 14.6% and 14.2% from 1989 to 2009, respectively (ROSSTAT, 2002, 2010). This period was also characterized by strong rural depopulation with a net loss of 27.5% from 1989 to 2010 (472,000 to 342,000 residents) in Ryazan Oblast and a similar decline in Mordovia Republic (−23.2%, 423,000 to 325,000 residents, Heaney, 2011; ROSSTAT, 2002). At the same time, relatively moderate urban population loss occurred (−7.6%, 876,000 to 809,000 urban dwellers, in Ryazan Oblast and −7.2%, 541,000 to 502,000 urban dwellers, in Mordovia Republic, Heaney, 2011; ROSSTAT, 2002). With a long history of land use, the study region is representative for European Russia, where heavy forest use started in the 18th and 19th centuries due to increased timber demand during industrialization. Forest management in Soviet times was characterized by overexploitation and forest resource degradation due to industrial pollution, and heavy exploration of the Asian part of Russia (Krankina and Dixon, 1992). At the time of the collapse of the Soviet Union, however, the Europe-Ural geographic region was still the center of timber production and consumption (Krankina and Dixon, 1992). The forests in the northern part of the study region mainly occur on marginal soils, and in the Northwest, the Meshchera Lowlands form a flat and marshy forested area that had been drained during the 20th century to enable peat extraction (Potapov et al., 2011). In the southern part, large-scale farming with row-crop agriculture is dominating, with livestock farming on the pastures in the floodplain areas of the Oka River and its tributaries. Because humans have exploited forests in European Russia for centuries, the present extent of intact forests in European Russia is small (Aksenov et al., 2002; Yaroshenko et al., 2001). This translates into a high priority to protect the remaining old-growth and close-to-nature forests, especially in the comparatively densely populated areas around Moscow.

Two strictly protected areas are located in the study region: Oksky State Nature Reserve and Mordovsky State Nature Reserve (Fig. 1). These zapovedniks were established in 1935 and 1936, respectively, and are characterized by intensive historical and current land use in their surroundings. Oksky State Nature Reserve was originally founded to protect the Russian desman (*Desmana moschata*) (Priklonsky and Tichomirov, 1989) and is located in the Meshchera Lowlands in the floodplain of the Pra River, a swampy area with poor, sandy soils that is part of a wetland of international importance (Oka & Pra River Floodplains, 1994, Ramsar Convention of Wetlands). The protected area covers about 77,000 ha of coniferous and mixed forests, wetlands, and meadows, and was designated as a biosphere reserve in 1978, with three protection zones of gradually differing intensities of permitted land use. The core zone (22,600 ha) equals the area of the zapovednik before 1989 and has the highest possible protection status (IUCN Ia). In the transition zone (33,100 ha; added in 1989), non-timber forest product use (e.g., collection of berries, mushrooms, and medicinal plants) is allowed. The buffer zone of 22,000 ha completes the biosphere reserve; there are few restrictions on land use in that zone (V. P. Ivanchev 2009, 2011, personal communication). Mordovsky State Nature Reserve is located about 130 km east of the Oksky State Nature Reserve over the area that had been protected by the Sarov monastery since the 18th century. Established to protect old-growth forests of the taiga zone (Tereshkin et al., 1989), it contains only one protection zone (IUCN Ia) encompassing 64,900 ha. Dominant land cover is coniferous and mixed forest, including some old-growth forest remnants (Tereshkin et al., 1989). The northern part of Mordovsky State Nature Reserve (22,400 ha) is a closed area and controlled by the city of Sarov, a Russian center for nuclear research.

2.2. Satellite images and ancillary data

We acquired a time series of 38 summer Landsat TM/ETM+ scenes covering three footprints of path/row 176/22, 175/22, and 174/22 for the years 1984–2010 (Table 1). Image availability was mainly limited by cloud coverage. Maximum cloud cover in the selected images was 22%. We excluded the thermal bands from further analysis due to their coarser resolution. We did not apply any radiometric normalization since the support vector machine (SVM) classifier should not be impaired by radiometric differences among images (Huang et al., 2002), and the forest disturbance index includes an image-based normalization procedure (Healey et al., 2005). We co-registered the nine images from the European Space Agency to the terrain-corrected L1T imagery from the United States Geological Survey (USGS) with a maximum positional error of <0.5 pixels (mean RMSE 0.347). To remove

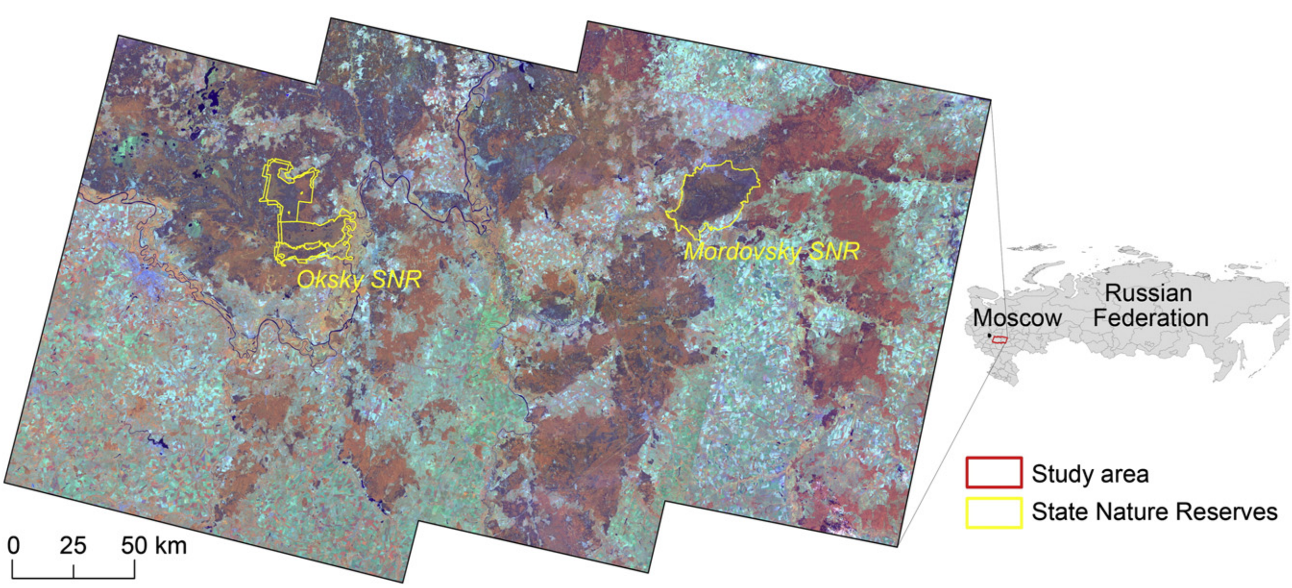

Fig. 1. Study area in European Russia with Oksky and Mordovsky State Nature Reserves (Landsat footprints path/row (acquisition date): 176/22 (2007-05-31), 175/22 (2000-05-28), and 174/22 (2007-08-21), in band combination 4/5/3, i.e., false color). Photosynthetically active vegetation is shown in reddish colors (e.g., different forest types or cultivated agricultural land).

clouds and cloud shadows, we manually digitized cloud masks on screen. We used an existing LULCC map for the Landsat footprint 176/22 for the time period 1988 to 2009 (Prishchepov et al., 2012a) as well as georeferenced topographic maps (1:100,000, VTU GSh, 1989). Furthermore, we used vector boundaries of the two protected areas (IUCN and UNEP-WCMC 2011; OSNR, 2009).

Several layers of biophysical and socio-economic variables were generated as covariates in the statistical analyses (see Section 2.4). First, the distance to forest edge, second, the distance to the nearest city (VTU GSh, 1989), third, the distance to the nearest road (VTU GSh, 1989), fourth, elevation (USGS Global Digital Elevation Model), fifth, slope (NOAA Global Land 1-km Base Elevation Project), and last, percent of evergreen trees versus deciduous (MODIS Land Cover, MCD12Q1, Land Cover Type 1 (2005): IGBP global vegetation classification scheme). Distances were calculated as Euclidean distances.

2.3. Change detection

Our mapping of land-use and land-cover changes in the study region incorporated two steps: (a) a SVM classification to map forestland and farmland abandonment, and (b) a trajectory analysis to determine forest disturbances (Fig. 2). Here, we define forest disturbance as the complete removal of tree cover in a Landsat pixel at a certain time, regardless of the cause, i.e., including both human-induced and natural forest disturbance.

First, we stacked images centered around 1988 and 2010 to derive a forestland mask (to be used in the trajectory analysis) and to map farmland abandonment. Detecting farmland abandonment is challenging due to the spectral complexity of this class (e.g., spectral ambiguities between intermediate crops as well as between fallow land and particular crops and grassland, young forest, and the great spectral variability in crop types before abandonment and post-abandonment succession vegetation). Capturing these phenological differences (e.g., varying stages of maturing and senescent crops in active agricultural land or low variation in abandoned fields with shrub encroachment) is important to separate active from abandoned agriculture (Baumann et al., 2011; Kuemmerle et al., 2008; Prishchepov et al., 2012b). Thus, we included two satellite images for each time step ideally acquired at different times in the growing season and in different years (path/row 176/22: 1988-07-21, 1988-08-22, 2007-05-31, and 2009-09-09; path/row 175/22: 1986-06-15, 1989-08-18, 2006-07-08, and 2009-07-16; path/row 174/22: 1987-06-11, 1988-07-23, 2007-08-21, and 2010-06-26).

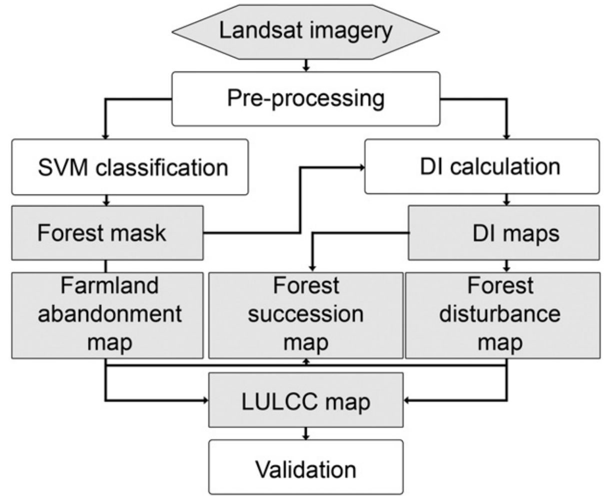

Fig. 2. Work flow of land-use and land-cover change (LULCC) detection (SVM = support vector machines, DI = disturbance index).

Table 1
Landsat imagery acquired for the years 1984–2010 (paths 176, 175, and 174; row 22).

Year	Path 176	Path 175	Path 174	Sensor
1984		25 June	06 September	TM-5, TM-5
1986	08 September	15 June		TM-5, TM-5
1987			11 June	TM-5
1988	21 July		23 July	TM-4, TM-4
1989	17 August	18 August		TM-5, TM-4
1991	24 September	03 October		TM-5, TM-5
1992	06 June			TM-5
1993		02 June		TM-5
1994	16 September			TM-5
1995	15 June	08 June	19 July	TM-5, TM-5, TM-5
1996		12 July		TM-5
1997	19 May			TM-5
1998	07 June		09 June	TM-5, TM-5
1999	06 September			ETM+
2000	14 July	28 May		TM-5, ETM+
2002	09 May	11 June	30 July	ETM+, TM-5, ETM+
2004			28 August	TM-5
2006	01 September	08 July	19 September	TM-5, TM-5, TM-5
2007	31 May	12 August	21 August	TM-5, TM-5, TM-5
2009	09 September	16 July		TM-5, TM-5
2010	24 June		26 June	TM-5, TM-5

We used support vector machines (SVM) as our classifier, a machine learning algorithm that is well suited to map spectrally complex classes (e.g., multimodal), which are common for change classifications (Huang et al., 2002). The basic approach of an SVM classifier is to identify a hyperplane that optimally separates two classes in the feature space. SVM frequently outperform other non-parametric and parametric classifiers (Foody and Mathur, 2004) and require few training data (Foody and Mathur, 2006). SVM have been successfully applied for mapping land-use change in general and farmland abandonment in particular (Hostert et al., 2011; Kuemmerle et al., 2008; Prishchepov et al., 2012a).

We classified the stack of four images for each footprint into five LULCC classes: (1) stable agriculture, i.e., arable fields and actively managed grasslands for hay cutting and livestock grazing that were in use in both points in time; (2) abandoned agriculture, i.e., fields and pastures that were in use at the end of the 1980s, but abandoned in 2010, including areas that had reverted to forests; (3) unmanaged grasslands and riparian trees; (4) forest, i.e., forest of different types as well as sites of forest disturbance and post-disturbance succession, but not abandoned areas; and (5) other including water, settlements, and roads. Training data were comprised of randomly distributed points (100–300 per class) that we labeled based on field visits (e.g., for farmland abandonment), very high resolution data provided via Google Earth, topographic maps (e.g., for elements of the 'other' class), and the Landsat satellite images themselves. Additionally, we digitized training points to bolster sample sizes for small and spectrally complex LULCC classes, such as disturbed forest areas (small) and farmland abandonment (spectrally complex).

Second, we applied a trajectory-based forest disturbance detection to map forest-cover change between 1984 and 2010. For each pixel of all images in our Landsat time series, we calculated the forest disturbance index (DI), which is a linear transformation of the normalized Tasseled Cap (TC) indices (Healey et al., 2005). The DI assumes that disturbed forest is characterized by high brightness, low greenness, and low wetness. First, the Tasseled Cap indices were normalized to a mean of zero and standard deviation of one using a representative forest reference population (i.e., all forest areas that remained undisturbed across time). Second, the normalized Tasseled Cap indices were linearly combined as: $DI = nBr - (nGr + nWe)$ where n refers to normalized Tasseled Cap (TC) brightness (Br), TC greenness (Gr), and TC wetness (We) components. In other words, normalized brightness values are reduced by the sum of the normalized greenness and wetness (Healey et al., 2005).

In a given DI map, a DI value of 0 specifies areas that are close to the mean spectral characteristics of forests and therefore likely denotes forest. Conversely, large DI values represent spectral dissimilarity to the forest reference population (e.g., a DI = 2 refers to a spectral dissimilarity of two standard deviations to the reference population), thus likely denoting non-forest areas (or a forest disturbance when analyzed in a temporal trajectory). Mapping forest disturbances requires setting two user-defined thresholds (Healey et al., 2005). A first threshold indicates the upper range of DI values of areas considered closed-canopy forests. The second DI threshold denotes DI values above which an area can be considered a non-forest (i.e., disturbed) area. Values between both thresholds characterize various stages of degraded or regrowing forest (Healey et al., 2005). We defined the two DI thresholds based on the DI statistics of areas with known disturbances in different forest types as well as undisturbed forest (based on field visits and visual digitizing from the Landsat images themselves) as well as experience from a range of previous applications of the concept in temperate forests (e.g., Healey et al., 2005; Kuemmerle et al., 2007) (Table S1). In our study area, DI values lower than 2 represented intact, undisturbed forest, whereas forest disturbances were characterized by DI values larger than 4 to larger than 10 (depending on the image). This variation was caused by differences in the phenological and weather conditions of our imagery over time, both affecting the Tasseled Cap indices. Another factor contributing to the variability in upper DI thresholds was the time interval between subsequent images in the time series that influenced the degree of post-disturbance forest succession on disturbance sites. Based on the two thresholds, we flagged each pixel in each image of our time series as either undisturbed or disturbed forest.

Once a time series of disturbance images was available, we carried out a trajectory analysis to remove false detections. When analyzing several DI maps in a temporal trajectory, an increasing DI value over time towards non-forest indicates forest disturbance, and a decreasing DI value over time characterizes forest recovery. Starting with a cloud-free image, we therefore identified those disturbance pixels that showed both a DI value lower than 2 in the first year and a value greater than the second threshold in the respective year of forest disturbance (Fig. 3).

To map forest disturbance, we used a minimum mapping unit of four Landsat pixels, i.e., 0.36 ha, which was chosen to sieve speckle and to remove pseudo-change pixels due to remaining positional inaccuracy of some images. We then visually checked all detected forest disturbances and evaluated whether a disturbance was caused by logging (e.g., regularly shaped, mainly rectangular form, mostly small) or fire (e.g., burn scar clearly visible in false-color combinations, irregularly shaped, often large), also using additional Landsat images, which were not included in the time series due to high cloud coverage. We labeled post-fire logging (i.e., clear-cutting on burned forest areas up to five years after the fire event, Schroeder et al., 2012) as logging since salvage logging represents forest management. This yielded annual forest disturbance maps for the period 1984–2010, where each forest disturbance was either labeled as logging or fire. We then calculated annual forest disturbance rates by dividing the area disturbed in a given year by the total forest area in 1984/86 (i.e., the forest mask from our initial SVM classification adjusted to the forest area in 1984/86 using the earliest images of our time series, path/row 176/22: 1986-09-08, path/row 175/22: 1984-06-25, and path/row 174/22: 1984-09-06). For years without image in our time series (Table 1), we evenly distributed the disturbance area mapped in the next year when an image was available across the observation year plus all preceding years in that gap period.

We then combined the land-cover change map and the forest disturbance map to assess forest succession both on abandoned farmland and in previously disturbed forests. Forest succession was mapped based on the similarity of a non-forest pixel to the mean spectral characteristics of forests (i.e., the DI value image). Specifically, we labeled abandoned areas as forests and disturbed areas as recovered once the DI values

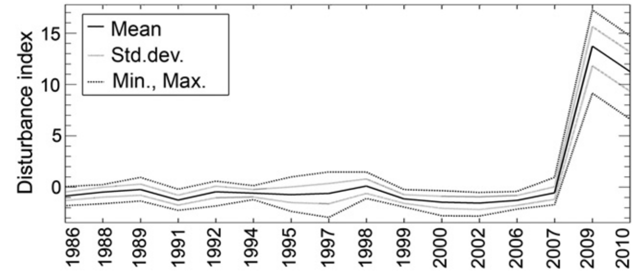

Fig. 3. Trajectory of forest disturbance index values across the available Landsat imagery (1986–2010, see Table 1) for digitized site (34 pixels) of a forest disturbance in 2009.

on these areas showed a DI value within two standard deviations around the mean DI of forest spectral characteristics ($-2<\mathrm{DI}<2$).

We validated our results based on a stratified random sample of points that was independent from those used for training. We used 300 points for each of the LULCC classes stable agriculture, abandoned agriculture, unmanaged grasslands and riparian trees, forest, and other, and 50 points for each of the 20 forest disturbance years (1986–2010). To minimize spatial autocorrelation, we used a minimum distance of 1 km between points. We labeled points based on very high resolution satellite images (available in Google Earth), the Landsat images themselves (Cohen et al., 2010; Kuemmerle et al., 2009; Zhu et al., 2012) and field visits. We calculated an error matrix, calculated user's, producer's, and overall accuracies, and corrected for sampling bias in the error estimates (Foody, 2002; Olofsson et al., 2013). We also calculated true area estimates as well as the 95% confidence intervals around these estimates based on the uncertainty in our LULCC map (Card, 1982).

2.4. Evaluating the effectiveness of Oksky and Mordovsky State Nature Reserves

To assess the effectiveness of the two strictly protected areas in preventing forest disturbances inside them during post-Soviet times, we first summarized forest disturbance both within the nature reserves (in case of Oksky State Nature Reserve separately for all protection zones) and in their surroundings. The latter was done using four buffers of 0–5, 5–10, 10–15, and 15–20 km from the outer boundary of the protected areas (Fig. 4). This represents the classic approach to measuring protected area effectiveness (Curran et al., 2004; DeFries et al., 2010).

Second, we evaluated the protected area effectiveness by using matching statistics to control for the non-random allocation of protected areas and the potential displacement of land uses to surrounding areas, e.g., forest disturbance spillovers to adjacent forests (Andam et al., 2008; Ferraro et al., 2011; Wendland et al., 2011). For our matching statistics, we took a random sample of 1% of forested pixels within the two protected areas and four times this number of forested pixels outside of the nature reserves. We then assigned each pixel a propensity score measuring the likelihood that the pixel was protected. A propensity score summarizes multiple characteristics into a single-index variable and is estimated using a logit model (Becker and Ichino, 2002). In total, only very few points within the forest areas (<1% for all sample sizes tested) were affected by fires. Forest fire is therefore negligible in our matching analyses and the observed effects of forest disturbances on protected areas effectiveness can be solely attributed to logging (including salvage logging). We included biophysical and socio-economic characteristics expected to impact the probability of protection in the propensity score. The distance to the nearest road served as a proxy for the impact of infrastructure, the distances to the nearest city and to Moscow served as a proxy for the importance of market access and outside timber demand (i.e., Moscow) (Mueller and Munroe, 2008; Wendland et al., 2011). Elevation and slope characterized the roughness of the terrain, thereby possibly affecting the effort of human-induced forest disturbance.

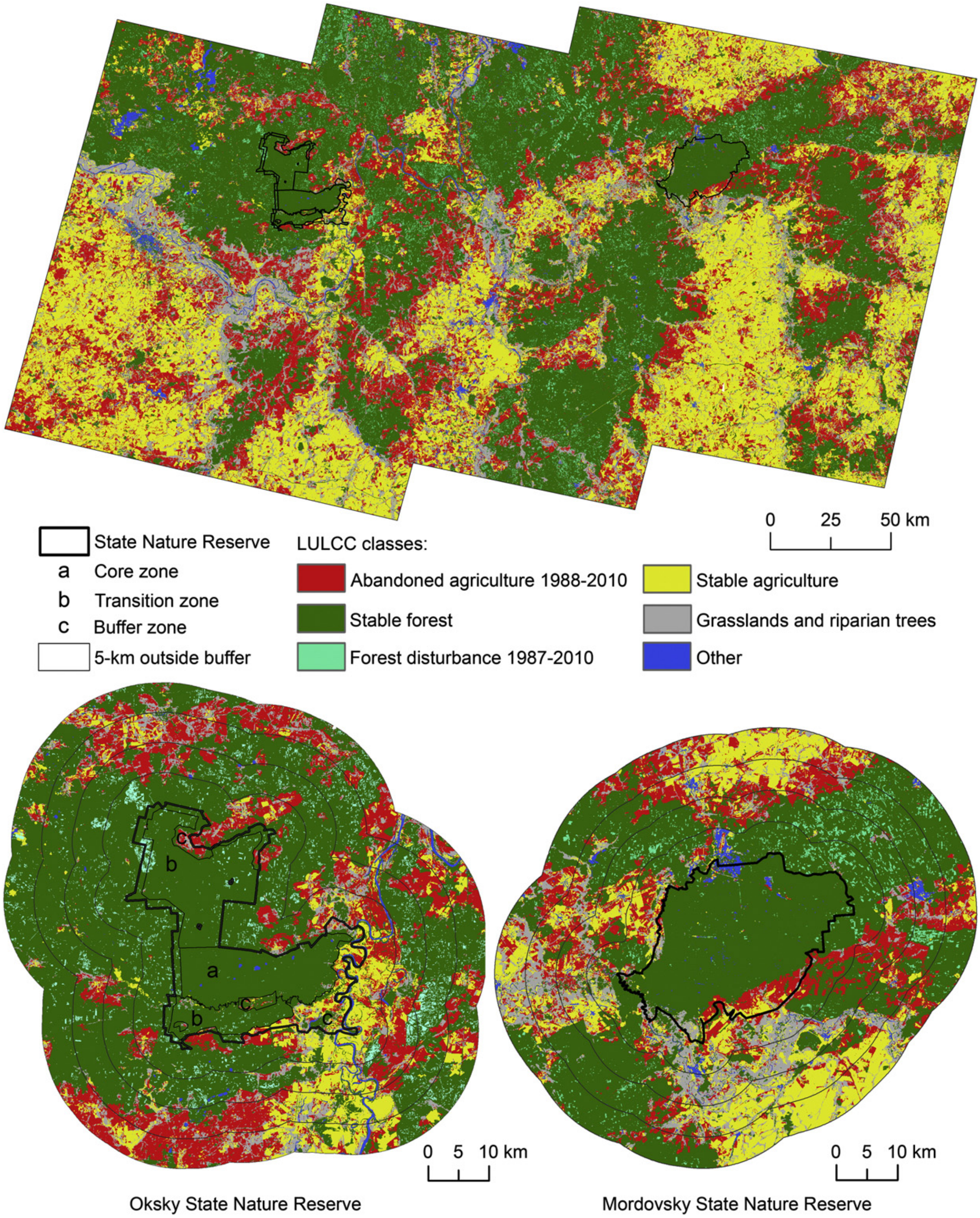

Fig. 4. Post-Soviet land-use and land-cover change (LULCC) within the study area and Oksky and Mordovsky State Nature Reserves with their surrounding ring-shaped buffers within 0–5, 5–10, 10–15, and 15–20 km of the outermost boundary of the protected areas.

The distance to forest edge is expected to indicate the impact of prior human-caused disturbance, and the percent of evergreen (versus deciduous) trees related to a potential influence of the forest type on the disturbance regime.

Observations within the protected areas were then matched to pixels outside based on the minimum linear distance between propensity score values. We dropped protected area pixels with a propensity score higher than the maximum or less than the minimum propensity scores of observations outside of the protected areas. Such "common support" ensures good matches (Caliendo and Kopeinig, 2008). The average difference in land-use outcomes was then calculated as the difference in means between these matched populations. However, matching does not always eliminate all differences between pixels within and outside of protected areas, and we checked for remaining differences by comparing the covariate balance in the matched sample. Covariate balance was calculated as the normalized difference in means:

Table 2
Confusion matrix for the merged LULCC map including the detected forest disturbances (B = background, A = stable agriculture, AA = abandoned agriculture, G = grassland and riparian trees, F = stable forest, YYYY = forest disturbance in respective year).

		Reference																								
		B	A	AA	G	F	1986	1987	1988	1989	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2002	2004	2006	2007	2009	2010	Total
LULCC Map	B	162	9	7	10																						188
	A	11	161	17	17																						206
	AA	2	27	146	16	1																					192
	G	6	10	18	148	2																					184
	F	7	1	7	16	176	2	7	3	7	3	2	4	3	3	6	9	6	7	6	6	6	4	2	3	2	298
	1986					1	35																				36
	1987							41																			41
	1988								45																		45
	1989	2			3					31					1												37
	1991	1									30																31
	1992											45															45
	1993												41														41
	1994													42													42
	1995	1													42												43
	1996					2										39											41
	1997					3											34										37
	1998	1				1												39						1			42
	1999																		43								43
	2000																			39							39
	2002				1																43						44
	2004				1																	42					43
	2006																						45				45
	2007				1																			39			40
	2009	2																							44		46
	2010																									47	47
	Total	195	208	195	213	186	37	48	48	38	33	47	45	45	46	45	43	45	50	45	49	48	49	42	47	49	1896

Table 3
Area-adjusted overall accuracy (OA), producer's (PA) and user's (UA) accuracies of the merged LULCC map including the detected forest disturbances (B = background, A = stable agriculture, AA = abandoned agriculture, G = grassland and riparian trees, F = stable forest, YYYY = forest disturbance in respective year).

	PA (%)	UA (%)
B	32.52	86.17
A	86.26	78.16
AA	75.95	76.04
G	61.89	80.43
F	98.98	59.06
1986	18.44	97.22
1987	20.37	100.00
1988	30.43	100.00
1989	8.68	83.78
1991	22.91	96.77
1992	10.37	100.00
1993	6.14	100.00
1994	10.71	100.00
1995	43.90	97.67
1996	1.53	95.12
1997	2.47	91.89
1998	8.22	92.86
1999	2.25	100.00
2000	7.99	100.00
2002	15.53	97.73
2004	3.98	97.67
2006	32.34	100.00
2007	18.52	97.50
2009	15.85	95.65
2010	27.50	100.00
OA (%) = 71.25		

$\overline{X_1} - \overline{X_2} \Big/ \sqrt{\sigma_1^2 + \sigma_2^2}$, where $\overline{X}$ is the mean covariate value, σ^2 the variance, and the subscripts designate areas within (1) and outside (2) of protected areas.

In general, a normalized difference in means greater than 0.25 is “large” (Imbens and Woolridge, 2009). When matching was incomplete, regressions of the matched sample were used to further reduce bias (Imbens and Wooldridge, 2009). We found that matching did not lead to complete covariate balance in our analysis; therefore, we performed a logistic regression analysis using the matched sample and controlling for each of the covariates listed above. The marginal effect (i.e., the derivative of the prediction function) of the protected area status on forest disturbance is equivalent to the effectiveness of the protected area because it describes the increase in likelihood of our outcome and thus reveals the mean effect of a protected area on forest disturbance. To enable a comparison of the effectiveness of Oksky and Mordovsky State Nature Reserves despite the differences in available satellite imagery for the various Landsat footprints across time (Table 1), we generated forest/non-forest maps for five 5-year time periods, i.e., 1986–1990, 1991–1995, 1996–2000, 2001–2005, and 2006–2010 and repeated the matching statistics for each time period.

3. Results

3.1. LULCC mapping

Our change analyses resulted in reliable maps of forest disturbance and farmland abandonment for the time period of 1984 to 2010. The area-adjusted overall accuracy of the LULCC map, containing 25 classes, was 71.25% (Tables 2 and 3). The most widespread classes stable agriculture, abandoned agriculture, and stable forest, were all mapped with moderate to high user's (all classes ≥59.06%) and producer's (≥75.95%) accuracies. The forest disturbance classes had on average high user's accuracies (mean = 97.19%), but lower producer's accuracies (mean = 15.41%) (Table 3).

In 2010, the study region was composed of a heterogeneous landscape characterized by 46.07% agricultural land (active and abandoned farmland), 40.20% forest area, 12.06% grasslands and riparian trees, and 1.67% water bodies, settlements, and roads (Fig. 4).

Abandonment of agricultural land was widespread in the study region and occurred on 18.37% of the total landscape in 2010, and 39.89% of the 1988 agricultural land (1,281,331 ha arable land in 1988) (Fig. 4). Most abandoned land was located in the vicinity of forests. On 9.20% of the abandoned area (117,897 ha) forests had established by 2010.

In our study region, 5.02% of the total forest area was disturbed between 1984 and 2010 (137,912 ha) (Fig. 4.). We did not find any repeated disturbances in our analyses. Annual forest disturbance rates varied from 0.13% in the years 1996, 1997, 1999, and 2000 up to 0.49% in the years 1985 and 1986 (mean 0.23%, standard deviation 0.1, Fig. 5). Our results also showed distinct temporal trends in forest disturbance rates. In the late Soviet era from 1986 to 1990, forest disturbance was highest (40,254 ha for the total period from 1986 to 1990, representing 1.44% of the total forest area in 1984). After the collapse of the Soviet Union, the disturbance rates declined to a low-point in the 1990s (29,338 ha of disturbed forest relative to the total forest area in 1984, equalling a share of 1.05% from 1991 to 1995, and 16,367 ha of disturbed forest equalling a share of 0.58% from 1996 to 2000). Subsequently, forest disturbance rates increased again, but only to about half of the rates of the late-Soviet period (23,187 ha and 21,279 ha from 2001 to 2005 and 2006 to 2010, respectively, equalling a share of 0.83% and 0.76%, respectively). Discriminating the forest disturbances due to fire and logging (including post-fire logging) reveals that the main trend in disturbances is due to logging (Fig. 5). Burned forest areas, however, increased markedly after 1999, sharing up to 21% in 2008 (Fig. 5). Although the accuracy of our forest disturbance classes varied over time (Table 3), the 95% confidence intervals of our area estimates were relatively moderate and did not suggest a bias regarding the overall trend in forest disturbance across the time period (Fig. 5).

Forest recovery on previously disturbed sites within the forest (i.e., not forest expansion on abandoned land) was also a widespread process in the study area. Our analyses suggested that forests required about 10–15 years to recover from disturbance and thus to be again classified as forest (Fig. 5). About 46.19% of the forest that had been disturbed between 1984 and 2010 had regrown by 2010 (63,708 ha).

3.2. Protected area effectiveness

Our matching statistics approach revealed that within both protected areas, Oksky and Mordovsky State Nature Reserves, forest disturbance rates were significantly lower than in their surroundings (Fig. 4). This suggests that both protected areas had a statistically significant effect on protecting forests inside them from forest disturbance.

In Oksky State Nature Reserve, forest disturbances occurred on about 1241 ha between 1986 and 2010, equalling 1.81% of the protected forest area (annual rate = 0.08%). Within the core zone of the protected area, only 0.19% of the forest area had been disturbed over that same period (41 ha), with annual forest disturbance rates never exceeding 0.03% (Fig. 5). Within the transition zone, disturbances were more frequent and occurred on 1.57% of the forest area in this zone between 1989 (i.e., the year of establishment) and 2010 (517 ha). Yet, within the first years after extending the protected area by the transition zone, annual disturbance rates decreased substantially from 0.56% in 1987 and 1988 (i.e., prior to establishment) to rates below 0.10% after 1993 within this area and remained at a low level. A sharp increase in disturbance rates due to several larger wildfires occurred after 2007, when rates exceeded 0.40% (Fig. 5).

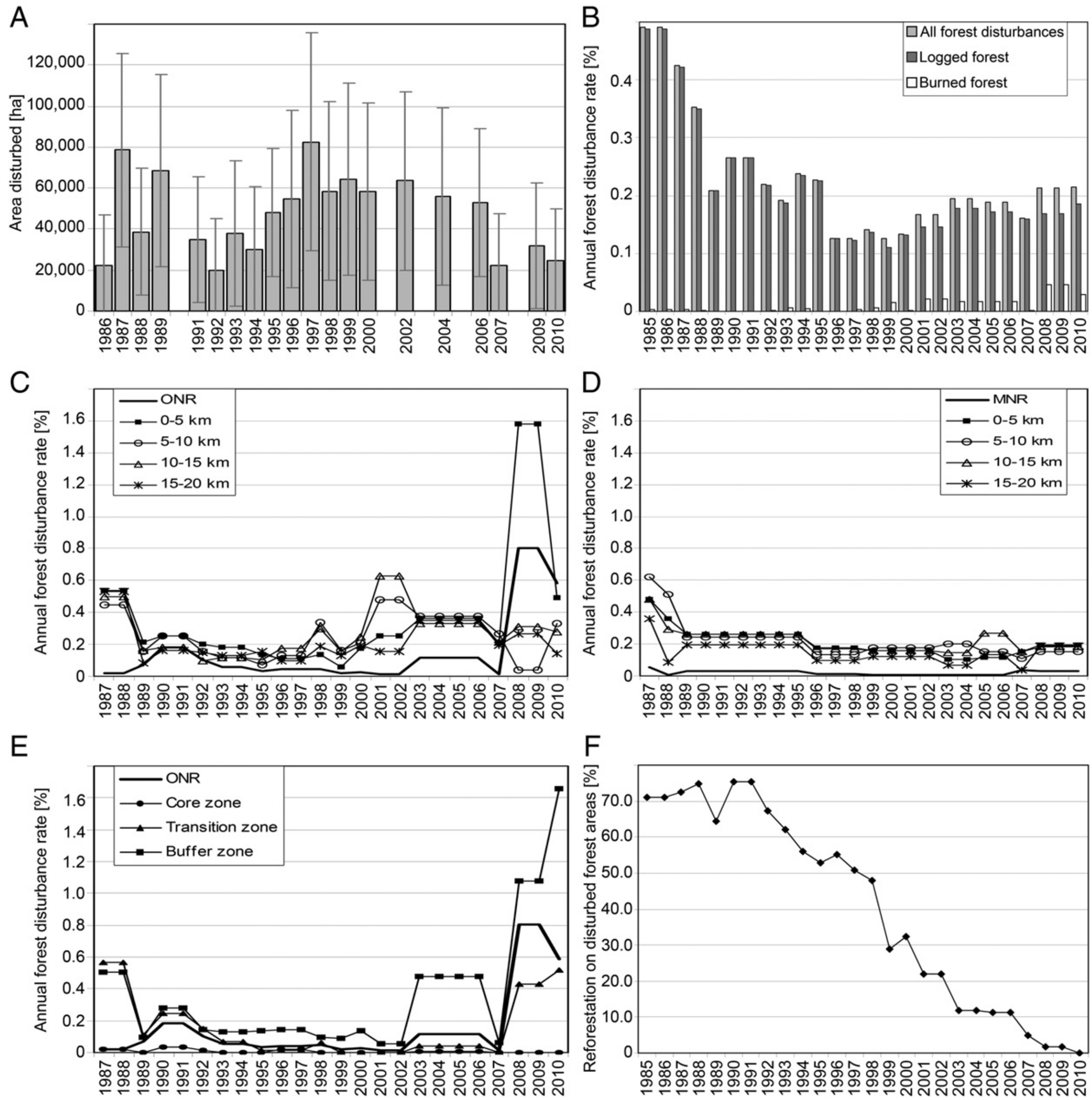

Fig. 5. (A) Annually disturbed forest area with error bars indicating the 95% confidence intervals; (B) annual disturbance rates (all forest disturbances, logged, and burned forest) for the entire study area in per cent of the total forest area in 1984/6; (C) annual forest disturbance rates for the protected areas of Oksky (ONR) and Mordovsky (D, MNR) State Nature Reserves and their surrounding ring-shaped buffers within 0–5, 5–10, 10–15, and 15–20 km of the outermost boundary of the protected areas; (E) annual forest disturbance rates for the different protection zones of Oksky State Nature Reserve (ONR); (F) reforestation on disturbed forest areas in the study region in 2010 (with year of forest disturbance).

Table 4

Matched and unmatched observations (percent (%) and number (N)) that experienced forest disturbance within Oksky State Nature Reserve (ONR), Mordovsky State Nature Reserve (MNR), and their outside areas (Controls); unmatched samples are indicating: N within the protected areas = 1% of forested pixels within the protected areas, and N Controls = 4× 1% of forested pixels outside of protected areas; observations were removed from the sample once forest was disturbed.

	1986–1990		1991–1995		1996–2000		2001–2005		2006–2010	
	Match	No match	Match	No match	Match	No match	Match	No match	Match	No match
ONR (%)	0.00	0.00	2.46	2.45	0.12	0.12	0.39	0.39	0.62	0.61
Controls (%)	1.94	1.12	3.75	4.10	0.64	0.74	1.52	1.88	1.06	0.95
ONR (N)	2220	2292	6862	6903	6716	6734	6711	6726	6610	6700
Controls (N)	2220	27,844	6862	27,532	6716	26,402	6711	26,206	6610	25,714
MNR (%)	0.20	0.20	2.19	2.15	0.05	0.05	0.86	0.84	0.05	0.05
Controls (%)	2.76	2.06	3.80	5.49	0.62	0.44	1.29	1.80	0.13	0.10
MNR (N)	5869	5881	5763	5869	5649	5743	5570	5740	5530	5692
Controls (N)	5869	23,523	5763	23,038	5649	21,774	5570	21,679	5530	21,288

Table 5
Relative probability (%) of an observation within Oksky State Nature Reserve (ONR) and Mordovsky State Nature Reserve (MNR) experiencing forest disturbance in comparison to being outside of the considered protected area in the respective time period (N= matched sample). A negative relative probability highlights that a forest pixel located within a protected area has a lower probability to experience forest disturbance than a forest pixel with the same characteristics outside the protected area. A forest pixel located within Oksky State Nature Reserve (ONR), for example, has a 2% lower probability of forest disturbance in 1986–1990 than a similar forest pixel outside ONR.

	1986–1990	1991–1995	1996–2000	2001–2005	2006–2010
ONR (%)	−2.0***	−0.9***	−0.6***	−1.2***	−0.3*
ONR (N)	2220	6862	6716	6711	6610
MNR (%)	−3.5***	−0.8**	−0.9***	−0.1	−0.1
MNR (N)	5869	5763	5649	5570	5530

Statistically significant at: ***1% level, **5% level, *10% level.

Within the adjacent buffer zone, disturbances occurred on about 5.09% of the total forest area (683 ha out of 13,415 ha; 1989–2010). Showing a similar pattern as in the transition zone, annual disturbance rates within the buffer zone remained below 0.15% in the late-Soviet era and until 2001, but increased to 0.45% by 2003 and further up to 1.66% in 2010 (Fig. 5). Outside of Oksky State Nature Reserve, forest disturbances occurred on about 4.80 to 6.15% of the forest area within each of the four 5-km buffer areas (1986–2010). Annual forest disturbance rates within the surroundings were always higher than within the protected area itself, except for the years after 2007, when large forest fires occurred within the transition and buffer zones of Oksky State Nature Reserve (Fig. 5). The forest disturbance trend of the surrounding area was similar to that of the entire study region, with annual disturbance rates decreasing in the 1990s and increasing after 2000. Disturbance rates after 2000 also varied more in magnitude from year to year and among the surrounding buffer areas than those before 2000.

The matching statistics analysis suggested that Oksky State Nature Reserve prevented forest disturbances inside its boundaries since the relative probability of a pixel experiencing forest disturbance within Oksky State Nature Reserve was lower compared to a pixel outside. This was true for all time periods we assessed, although among time periods, probabilities varied and increased in general (from −2.0% in 1986–1990 to −0.3% in 2006–2010, Table 5). Despite the overall relatively low probability of forest disturbance in the study region (varying probability, but in general declining from 1.12% in 1986–1990 to 0.95% in 2006–2010, Table 4), and although very low rates of forest disturbance occurred within Oksky State Nature Reserve, the forests in the protected area were much less disturbed than forests outside the reserve.

Regarding Mordovsky State Nature Reserve, we found similar patterns. Within the protected area, about 277 ha of forest (0.50%) were disturbed in the period from 1986 to 2010. Annual forest disturbance rates were very low at all times (mean 0.02%, standard deviation 0.02, Fig. 5). In the four 5-km buffers outside of Mordovsky State Nature Reserve, disturbances occurred on 3.54 to 5.21% of the forests in the period from 1986 to 2010. Here, annual disturbance rates were about 0.50% in the late Soviet era, decreased to <0.26% in the 1990s, and remained below 0.20% after 2000 (Fig. 5). All 5-km buffers outside of Mordovsky State Nature Reserve always exhibited higher annual forest disturbance rates than the protected area itself (from 0.05 times higher in the 20-km buffer in 2007 up to 89.51 times higher in the 10-km buffer in 1988).

The matching statistics again revealed the effectiveness of Mordovsky State Nature Reserve, similar to Oksky State Nature Reserve. The relative probability of a pixel experiencing forest disturbance within Mordovsky State Nature Reserve was always lower than outside, although it increased from −3.5% in 1986–1990 to −0.1% in 2006–2010 (Table 5). This confirms that the protected area was effective in preventing forest disturbance inside its boundaries.

4. Discussion

4.1. Post-Soviet land-use changes

Our analyses revealed substantial and widespread LULCC in the post-Soviet era in our study region in European Russia, most importantly widespread forest disturbance due to logging as well as farmland abandonment and subsequent reforestation. Protected areas in our study region remained effective in the post-Soviet period in safeguarding their forests from human-caused disturbance. This is remarkable, considering the institutional instability and economic hardships of the transition period from state- to market-oriented economies, and stands in contrast to protected area effectiveness elsewhere. Our results therefore provide hope for conservation during turbulent times and they provide an example of how combining Landsat trajectory analyses and matching statistics can help to monitor the success of conservation.

Changes in forest cover exhibited distinct spatial and temporal patterns, particularly the initial decline of forest disturbance rates in post-Soviet times accompanied with an increase in forest cover on former agricultural land. The initial decline of disturbance rates was most likely caused by the crisis of the forestry sector during the transition of the state-planned Soviet economy to a market-driven economy, due to the slowly developing institutional framework for the forestry sector and lacking investment incentives (Torniainen and Saastamoinen, 2007). Following this initial contraction, demand for timber increased again leading to rising exports after 2000, which in turn spurred logging rates (Baumann et al., 2012; Henry and Douhovnikoff, 2008; Potapov et al., 2011; Wendland et al., 2011). Part of the increase in disturbance rates we observed after 2000 is also due to natural disturbances, particularly fires, which have been increasing in the study region during that time. Several larger wildfires, for example, in 2002, but especially after 2007, caused extensive forest loss and wildfires following severe droughts affected in particular the drained forested peatlands in the Meshchera Lowlands in the North of the study area. We note that while these fires were extensive, disturbances due to logging were dominating in our study area. Both types of disturbance affect forest ecosystems, yet there are considerable differences in vegetation structure, community composition, natural vegetation recovery, soil properties, and landscape fragmentation and connectivity after logging or fire disturbances (Lindenmayer and McCarthy, 2002; Lindenmayer and Noss, 2006). Our results of post-Soviet land-use changes confirm earlier studies in other areas of Eastern Europe. The initial decline of forest logging rates was widespread in Russia (Peterson et al., 2009). The disturbance rates for our study region were even below those in other regions of post-socialist Eastern Europe, for example, Ukraine, Slovakia, and Romania (Griffiths et al., 2012; Knorn et al., 2012; Kuemmerle et al., 2007), which is surprising given that our study region was relatively close to Moscow, Russia's major market center. During socialism, forests were overexploited in many regions across the Soviet Union (Nijnik and van Kooten, 2006), but whether the lower timber harvesting rates since 2000, which are only about half of the Soviet rate in our study region, are more sustainable, remains unclear. Old-growth forests in that region of Russia are scarce (Yaroshenko et al., 2001). Timber extraction is still a main threat to Russian forest habitats and protected areas (Ervin, 2003), and illegal logging continues (EEA, 2007; Tyrlyshkin et al., 2003).

The abandonment of farmland in post-Soviet time was the most widespread land-use change in our study area. The main underlying causes of abandonment in Russia were the breakdown of Russia's agricultural sector due to disappearing, formerly guaranteed markets for agricultural products and timber within the Soviet sphere of influence, price liberalization of agricultural inputs (e.g., fertilizer, machinery) and outputs (e.g., agricultural products) due to the deregulation of fixed market prices, rising international competition, a shortage of labor in Russia's rural areas due to outmigration into urban areas accompanied with low birth rates and a decreasing life expectancy during the 1990s, and the post-Soviet reforms in land ownership and market structures (Brooks and Gardner, 2004; Ioffe et al., 2004; Lerman et al., 2004; Prishchepov et al., 2012a). The high rate of abandonment in our study area was similar to abandonment rates in other regions in European Russia (Prishchepov et al., 2012a), and ranks among the highest in Eastern Europe (Baumann et al., 2011; Kuemmerle et al., 2008; Prishchepov et al., 2012a). Recultivation of agricultural land has increased in Russia since 2005, and a strong interest in Russia's currently unused land for producing both food and bioenergy is arising (Vuichard et al., 2008). Yet, this was not the case in our region, where abandonment continued to increase in the second post-Soviet decade and the rate of recultivation of abandoned farmland was generally low (1.5% in Ryazan Oblast, 2000–2010, Prishchepov et al., 2012a). Abandoned farmland typically transitions to grassland and then to forest via several successional stages. In our study area, many abandoned farmlands (>10.0%) had already reverted to forests and it is unlikely that these lands, particularly those on poor soils, will be put back into production due to limited interest in such land and high recultivation costs (Larsson and Nilsson, 2005; Schierhorn et al., 2012). Forest succession on abandoned marginal farmland will likely continue in the near future, affecting landscape configuration and forest connectivity. Currently, these post-agricultural forests are not managed by the forest service. Although the future of abandoned farmlands remains unclear (some recultivation has recently been taking place on fertile land in our study region), an appropriate management of abandoned land would lower the risk of exacerbating fires originating from these lands with unmanaged forest succession that may impact both biodiversity and ecosystem services (Navarro and Pereira, 2012).

Although our change detection approach yielded reliable maps of post-Soviet LULCC for our study area in European Russia, some uncertainties remain. First, our forest disturbance estimates are likely conservative due to the minimum mapping unit we applied and the relatively high disturbance index thresholds we used, which were selected to minimize errors of commission of the forest disturbance class (Lu et al., 2004). Second, while we visually classified natural and fire disturbances to assess general trends in these disturbance causes, we did not identify the causes of disturbance in our change detection. If training data on different types of disturbances would become available, a more comprehensive assessment to discriminate natural and human-induced forest disturbances could be performed (Schroeder et al., 2011). Third, we chose a conservative approach to assess succession on abandoned farmland as well as forest recovery of disturbance sites via labeling only those pixels as reforested that were spectrally similar to mature forest. This may have resulted in an underestimation of forest expansion and forest recovery as early-successional forest often lacks typical shadow effects in mature forests and is composed of different tree species (e.g., *Betula* or *Pinus*) with brighter reflectance than mature forests (i.e., leading to higher DI values). Fourth, some of our forest disturbance estimates had low producer's accuracy. A visual assessment suggested that wrongly classified validation points were mainly due to remaining positional inaccuracy among the USGS L1T and the images from other sources. Although individual positional accuracy was high (<0.5 pixels for all images), co-registration errors caused the misclassification of a few points, in particular, at the edge of forest disturbances (Zhu et al., 2012), which often was classified as undisturbed forest. As these omission errors were found in relatively small classes, the area weighting we carried out penalized such misclassifications strongly. It is important to note though that these accuracies have no significance for any of our conclusions since they mainly represent misregistration errors which are likely not biased towards a certain time period or area within our study region. Furthermore, we incorporated the uncertainty in our analyses by calculating true area estimates for all classes as well as the 95% confidence intervals around these estimates. We also note that our error estimates and change rates are well in line with other studies (e.g., Baumann et al., 2012; Potapov et al., 2011; Prishchepov et al., 2012a).

Changes in land use and land cover occurred in our study area throughout the entire period of 1984 to 2010, with forest disturbances and farmland abandonment likely affecting habitat availability and fragmentation for a variety of species. Only time will tell, however, what the exact effects of current trends of post-Soviet LULCC at the species level are, and whether these trends will continue into the future. Generally, current LULCC trends may pose both threats and opportunities. For example, continued abandonment of farmland could lead to widespread forest expansion, benefitting those species thriving in natural ecosystems (Kuemmerle et al., 2010; Orlowski, 2010). Conversely, farmland abandonment may threaten agrobiodiversity (Fischer et al., 2012), as highlighted in the Carpathians, for example, where abandonment threatens subalpine grasslands (Baur et al., 2006). Moreover, accelerating forest logging rates and recultivation of fallow land (with intensified agricultural use) in the surroundings of the nature reserves may pose new challenges for conservation and protected area effectiveness. Further research is needed to assess future threats and opportunities for conservation in the temperate broadleaf and mixed forest biome, one of the currently most threatened biomes in the world (Hoekstra et al., 2005).

4.2. Effectiveness of Oksky and Mordovsky State Nature Reserves

The two strictly protected areas, Oksky and Mordovsky State Nature Reserves, were overall effective in limiting logging within their boundaries, despite the rapid institutional changes after the breakdown of the Soviet Union. This is surprisingly, given that some protected areas in Russia were struggling after the breakdown of the Soviet Union (Colwell et al., 1997; Fiorino and Ostergren, 2012) and nature reserves in other post-socialist countries, for example, the Ukraine (Kuemmerle et al., 2007) or Romania (Ioja et al., 2010; Knorn et al., 2012), were less effective in preventing threats to habitats and wildlife. The reasons for this remain unclear, but potential explanations are the long time period that both protected areas existed, the relative closeness of both protected areas to Moscow, the fact that they are federally managed by the Ministry of Natural Resources and Environment of the Russian Federation, the comparatively numerous and well-trained protected area staff, or the reason that funding may have declined less for these protected areas than for others in Russia (e.g., Oksky State Nature Reserve is a major center of crane and European bison breeding and participates in many international projects). Further explanations are the relatively low population density in the study region, the ease of access to similar forest resources outside the protected areas, as well as the reduced pressure on the forests due to the generally decreasing forest disturbance rates in post-Soviet times.

Over time, the effectiveness of our two protected areas on curbing forest disturbance declined, but this was largely because the probability of forest disturbance in their surroundings declined in the post-Soviet period as well (Wendland et al., 2011). Thus, in terms of forest disturbance, the surroundings of the protected areas became more similar to the protected areas themselves (Tables 4 and 5). Our results also highlighted the lagged effect that the establishment of protected areas can have in terms of effectiveness. We detected only very small forest disturbances within the core zone of Oksky State Nature Reserve during 1986 to 2010, but most forest disturbances occurred in the transition and boundary zones, especially in the years immediately after the collapse of the Soviet Union due to human-induced forest clearances. Although these zones officially had been part of Oksky State Nature Reserve since 1989, the transition zone was not fully implemented until 1995, and this is reflected in the higher disturbance rates in our results. Mordovsky State Nature Reserve has also limited forest disturbance within its boundaries. Most disturbances were detected within the closed zone controlled by the city of Sarov; however, the remaining area that was managed by the protected area staff was effective in restricting forest disturbances.

Though the causes of natural and human-induced disturbances are different, there are linkages between the two disturbance types in our study area. Importantly, we found salvage logging to occur after forest fires. For example, fire events in the buffer Zone of Oksky State Nature Reserve triggered forest management and our results showed an increasing trend in fire events since 2000. This could have resulted in an increase in salvage logging within the protected areas and their surroundings.

Our analyses also highlighted that post-Soviet land-use change fundamentally restructured the surroundings of protected areas and thus, was impacting the “zone of interaction” the protected areas are embedded in. Although we detected post-Soviet LULCC such as forest disturbances and farmland abandonment (e.g., the abandonment of meadows that were used for hay cutting in Soviet times within the buffer zone of Oksky State Nature Reserve, V. P. Ivanchev 2009, 2011, personal communication) within the protected areas, LULCC was far more extensive in their surroundings. While these LULCC trends likely affect landscape configuration, further research is necessary to quantify these changes. Forest fragmentation is promoted by forest disturbances and, at the same time, by the large-scale forest succession on abandoned farmland that, even in the vicinity of the protected areas, provides the opportunity to increase forest cover and to establish novel connections between protected and unprotected species habitats.

Both methods of effectiveness estimation that we applied, the descriptive inside–outside comparison and the matching comparison, yielded relatively similar results. Yet, simple buffers, the traditional method to estimate protected area effectiveness, would not have allowed for the detailed picture provided by the matching statistics (e.g., quantification of the effect of protection, assessment of changes in effectiveness over time).

5. Conclusion and outlook

The rapid institutional and socio-economic changes following the breakdown of the Soviet Union in 1991 triggered a drastic episode of land-use and land-cover change in our study area in European Russia. Using a time series of Landsat TM/ETM+ images, we found strong changes in logging regimes as well as widespread farmland abandonment with extensive forest succession, which likely were triggered by the fundamental socio-economic and institutional transformations. The post-Soviet period was also characterized by institutional decay, diminishing funding, and a lower level of control and this brought about substantial challenges for nature conservation in Russia. Here, we showed that despite these challenges the two strictly protected areas we assessed, Oksky and Mordovsky State Nature Reserves, remained relatively effective in safeguarding their territory from logging during the period from 1987 to 2010. This confirms that these protected areas are not “paper parks” (Bruner et al., 2001). Even during the turbulent years after the breakdown of the Soviet Union, these protected areas had a measurable effect, highlighting the importance of protection efforts. Our results thus also contribute to the wider discussion of what determines the success of protected areas providing an encouraging example that protection can work in regions of the world that are undergoing socio-economic or institutional shocks. Rapid LULCC, however, occurred within the “zone of interaction” (DeFries et al., 2010) of both nature reserves, restructuring the wider landscapes the protected areas are embedded in.

For the future, recent LULCC trends may pose both threats and opportunities for nature conservation. Threats include continued or increasing logging resulting in increasing habitat fragmentation, the spread of fires from abandoned farmland where forests are unmanaged, or recultivation of currently unused lands, whereas opportunities could rise where forest expansion on former farmland increases habitat availability and connectivity. Predicting socio-economic shocks such as the breakdown of the Soviet Union is difficult or impossible and one reason for the wide range of plausible outcomes in future biodiversity scenarios (Pereira et al., 2010). This emphasizes the need for continued monitoring of protected areas within the larger landscape they are embedded in, and combining remote sensing with matching statistics is a promising avenue for doing so.

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