Understanding the long-term dynamics of Forest Transition: From deforestation to afforestation in a Mediterranean landscape (Catalonia, 1868-2005)

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Highlights

- Forest transition in Catalonia experienced two different phases from 1856 to 2005
- Current forests are less resilient and subjected to a new fire regime
- Reconstructing land-use dynamics is a useful knowledge to improve the forests

Understanding the long-term dynamics of Forest Transition: From deforestation to afforestation in a Mediterranean landscape (Catalonia, 1868-2005)

1. Introduction

Sustainable forest management aimed at enhancing supporting and regulating ecosystem services, together with lumber and firewood, can take advantage of a deeper knowledge of the long-term dynamics of forest areas. Assessed either by changes in land use practices or in land cover patterns (Roesch and Van Deusen, 2012; Coulston et al., 2014), their historical legacy of disturbance regimes strongly affects the properties of forest landscapes in terms of forest structure, biodiversity, and all sorts of ecosystem services that shape their actual possibilities at present (Foster et al., 2003; Millennium Ecosystem Assessment 2005; Schröter et al. 2005; Turner et al., 2008; Angelstam et al., 2011; Costa et al., 2014; Morrissey et al., 2015; Basnou et al. 2016). Recovering the long-term socio-ecological history behind woodland patch units is an important tool for carrying out active ecological restorations (Honnay et al., 2004; Parrotta et al., 2006; Jackson and Hobbs, 2009; Inger et al., 2015). It requires new interdisciplinary approaches and methods able to advance in some on-going theoretical, empirical and political debates on how patch dynamics of land-use change drives global change and affects ecosystem services. These debates include the Intermediate Disturbance Hypothesis, long discussed in Ecology (Shea et al., 2004; Roxburgh et al., 2004; Svensson et al., 2012; Pierce, 2014) but seldom applied to forestry and agroecosytems (Marull et al., 2015a, 2016a), and the controversy between land-sparing and a land-sharing approaches to biological conservation (Green et al., 2005; Fischer et al. 2008; Tscharntke et al., 2012a). A main challenge is understanding biocultural complexity in human-nature interaction at landscape level, which requires overcoming a view of nature deprived of human intervention and the frequent confusion of land abandonment with its 'rewilding' or 'renaturing' (MacDonald et al., 2000; Agnoletti and Rotherham, 2015; Rotherham, 2015).

These debates include the positive and negative views on the environmental impacts of current forest transition mainly driven by cropland and pastureland abandonment (Rey Benayas et al., 2007, 2009; Bullock et al., 2011; Queiroz et al., 2014; Plieninger et al. 2014; Otero et al., 2011; Otero et al., 2015; Marull et al., 2015b). Although deforestation continues being a main process at global scale, a significant reduction in the net forest loss has taken place during the last decades due to the so-called forest transition (FT) (Meyfroidt and Lambin, 2011; Pagnutti et al., 2013). FT is defined as the shift from contraction to expansion of woods, when nations attain an advanced level of economic development and their raw materials provision tends to be shifted from biotic to abiotic materials and displaced towards other countries' extraction (Rudel et al., 2005; Bruckner et al., 2012; Walker, 1993, 2012). In North America and many European countries FT began at the end of the 19th or the early 20th centuries

(Foster et al., 1998; Kauppi et al., 2006), whereas in some regions of Eastern Asia and Latin America has only recently started (Mather, 2007; Lambin and Meyfroid, 2010). In the Mediterranean region FT has taken place in the last 50-70 years, after a deforestation peak attained during the first half of the 20th century (Grove and Rackham, 2001) following strong socioeconomic forces (Casals, 2005; Pèlachs et al. 2009; Iriarte-Goñi, 2013; Cervera et al., 2015) which led to depopulation and deep socioecological changes in mountain and steep areas (Falcucci et al. 2007; Marull et al. 2010, Basnou et al. 2013).

The environmental impacts of deforestation are widely recognized, but FT also entails noticeable ecological effects not yet well known, such as the change of fire regimes or the loss of habitats in increasingly homogeneous agro-forest landscapes (Loreau et al., 2003; Flinn et al., 2005; Malhi et al., 2008; Culas, 2012; Tscharntke et al., 2012b). The unmanaged regrowth of woodland biomass has resulted in increasing wildfires (Lloret et al., 2002; Marlon et al. 2008; Loepfe et al., 2010, 2011; Pausas and Fernández-Muñoz, 2012; Nunes, 2012; Fernandes et al., 2014). In turn, these wildfires are leading to an intermittent rise of post-fire harvesting of salvaged wood. We hypothesize that these woodlands are caught in a vicious circle: forest abandonment is fuelling wildfires, which lead to greater post-fire wood removals. This trap comes from a deeper socio-ecological dynamic: the lack of a proper forest management keeps these regrowth woodlands in a low ecological state, rendering them less resilient to withstand a set of ever growing disturbances. In addition to wildfires, these sort of recurrent events include fallen trees after episodes of strong winds, snow or drought. The forced harvesting of fallen or burnt trees after these seemingly 'natural' disturbances well might be considered a sort of 'fitful' or 'spasmodic' forestry. We will see below its importance in Catalonia during the last decades, by analysing forestry series confronted with the chronology and scope of wildfires.

At the same time, the vanishing of agro-forest and wood-pasture mosaics may lead to a biodiversity loss that endangers landscapes' ecosystem services and biocultural heritages (Petit et al., 2001; Stoate et al., 2009; López-i-Gelats et al., 2011; Basnou et al., 2013; Agnoletti and Rotherham, 2015). This is likely to happen when a previous large wave of deforestation, driven by cropland expansion and overexploitation of woods, has given way to a fast encroachment of a young and homogeneous woodland cover in former cultural landscapes left abandoned, while the remnants of truly mature forest units become too scarce. Even more, the two main side effects of FT, wildfires and the decrease in landscape complexity, may interact one another and with the lack of sustainable forest management (Marull et al., 2016b). Large and frequent wildfires, like other seemingly 'natural' disturbances, may sometimes prevent recent afforested areas from becoming mature, complex and diverse. Under these circumstances, the new habitats offered within mature woodland units cannot compensate for the loss of many inner and edge habitats formerly provided by patchy cultural mosaics that are currently disappearing (Sirami et al., 2010; Geri et al., 2010;

Loepfe et al., 2010; Li et al., 2011; Miranda et al., 2015). To bring to light these unintended effects, more studies are needed on socioeconomic drivers and ecological outcomes of FT. This article aims to contribute to this line of research in the Western Mediterranean region from a long-term historical perspective.

Previous works suggest the need to overcome a simple 'FT/no FT' dichotomy, and highlight the complex and diverse site-specific trends of forest Land-Use Land Cover Change (LULCC) experienced depending, among other things, on landownership structure and public policies (Redo et al., 2009; Schulz et al., 2010; Chowdhury and Moran, 2012; Lindström et al., 2012; Sheffer, 2012; Tavares et al., 2014; Carmona and Nahuelhual, 2012; Devaney et al., 2015). Relevant information to understand causes and consequences of FT lies in the interaction between socioeconomic drivers and environmental conditions. For example, a number of authors point out to the dichotomy of economic development *versus* forest scarcity trends (Rudel et al. 2005, Meyfroidt & Lambin 2011) to explain the distribution of new forests resulting from FT compared with the pre-existing ones. In many cases, farmers abandon their more remote, less productive fields and pastures in search of better paid jobs due to urban and industrial development (Mather, 1992). In other cases, however, the loss of forests during cropland expansion may raise the prices of forest products, thus inducing landowners to plant trees in highly productive and accessible lands instead of using them for cropping or pasture. These alternative ways driving FT might simultaneously occur in different locations, or even alternate one another along time, depending on changes in the price of forest products, labour expenses, and travel costs to urban places of consumption. The former case seems more frequent in Mediterranean mountains, where forest regrowth following rural abandonment has predominated (e.g. Basnou et al. 2013). At larger scales, internal migrations from rural to urban areas have led to a double process of land-use change in Mediterranean regions: abandonment of traditional uses in highlands followed by woodland encroachment, and land-use intensification in lowlands (Gerard et al. 2010; Parcerisas et al. 2012; Basnou et al 2013). All these factors may have determined changes in the spatiotemporal paths of FT, whose actual patterns still remain largely unknown.

This article aims at identifying the main socioeconomic drivers and ecological consequences of the LULCC carried out in the forest covers of the study area. Section 2 describes the study area and methods used. Section 3 presents the results. The discussion carried out in section 4 focuses on the combination of anthropogenic causes and ecological consequences found in our results when they are framed in a historical perspective (1868-2005). This long-term standpoint highlights the importance of land-use legacy behind the current state of different forest units in terms of age composition, structure, maturity and resilience capacity in front of growing human and natural disturbances—such as wildfires, droughts or increasingly sporadic cuts (Sheffer et al., 2012; Salvati and Ferrara, 2013; Salvati et al., 2013). We conclude in section 4 that the present ecological state of these forests is a result of their long-term dynamics as cultural landscapes, and knowing better their history can help to design better forest management and land-use planning aimed at an active ecological restoration (Honnay et al., 2004; Parrotta et al., 2006; Jackson and Hobbs, 2009; Agnoletti, 2014; Tello et al., 2006, 2014).

2. Study area and methods

2.1 Study area

The study has been performed in central Catalonia (Fig 1), along a section of the North-South axis following the Llobregat River that runs from the Pyrenees to a small Mediterranean delta near Barcelona. It comprises almost all the present Berguedà and Bages counties where forest land cover predominates. Throughout these 2450 km^2 the topographic gradient ranging from 150 to 2500 m.a.s.l. creates a bioclimatic contrast between the agro-forest mosaics in the lowland plain surrounding the town of Manresa in the Bages County, and the forest-meadow mosaics in the steep mountains over the town of Berga. Currently, 58% of land corresponds to forestland. This includes: Mediterranean forests dominated by *Pinus halepensis*, *Pinus nigra*, *Pinus pinea*, *Quercus ilex*, and *Quercus x cerrioides*; temperate forests dominated by *Pinus sylvestris and Fagus sylvatica*; and subalpine forests dominated by *Abies alba* and *Pinus uncinata*.

In 1787 the lowland Bages County had 30,874 inhabitants and a population density of 23.8 inhab./km^2, whereas in the Pyrenees the Berguedà County had 16,292, and 13.7 inhab./km^2. In 1857 population almost doubled up to 58,852 (45.4 inhab./km^2) and to 31,759 (26.8 inhab./km^2) respectively, fostered in the Bages by vineyard expansion and the industry of Manresa city, and in the Berguedà by the urban growing demand of meat, wood, charcoal and other raw materials coming from forests. Then in 1900 population only slightly grew up to 67,381 in the Bages County (52 inhab./km^2), due to the *Phylloxera* plague that hit all vines (Badia-Miró et al., 2010), and shrunk 14% to 27,217 (23 inhab./km^2) in the Berguedà County. In 1960 population had almost doubled again in Bages up to 127,718 (98.6 inhab./km^2), while in Berguedà grew 76% up to 47,953 (40.5 inhab./km^2). In 2006 Bages had a population of 173,236 (133.8 inhab./km^2) mainly devoted to industry and services, whereas in Berguedà had shrunk 16.5% to only 40,064 (33.8 inhab./km^2).

These general trends on population densities markedly differed according to altitude and proximity to cities. Average altitude of villages with decreasing population trends from mid-19th century onwards was 514 m a.s.l. in the Bages County (against 322 m a.s.l. in growing localities), and 958 m a.s.l. in the Berguedà County (against 655 m a.s.l. in growing ones). From the 1950s onwards population tended to concentrate in the main towns and small cities which in most cases kept growing, but based in industrial activities in the lowlands around Manresa city, whereas it decreased in many small mountain villages more dependent on the exploitation of natural resources. In the higher lands of the Pyrenees depopulation has only been halted and even reversed thanks to tourist activities (Fig 1).

Figure 1. Population growth and decrease in the two Catalan counties of the study area, 1787-2014. Dots are census data, and lines are interpolations.

From mid-19th century up to the 1950s the demand and price of forest raw materials remained high because new industrial and urban uses of wood tended to be added to traditional ones, rather than replacing them (Sala, 2003; Iriarte-Goñi, 2013). This stage corresponded to the development of the First Industrial Revolution in Catalonia and Spain, which was fuelled by coal, and fossil fuels were supplementing many traditional biomass uses instead of superseding them (Kander et al., 2013; Infante-Amate et al., 2015). It was not until the second half of the 20th century that fossil fuels and lumber imports started to replace domestic forestry products, fostered by cheaper international transport freights during the new oil era that fuelled the Second Industrial Revolution as well as the Green Revolution in farming (Nadal et al., 2012; Naredo, 2004). The use of feed imports to fatten livestock in feedlots led to the end of the old Mediterranean transhumance and other types of extensive animal husbandry (Ruiz and Ruiz, 1986; Bunce et al., 2004; Fernández-Giménez and Fillat-Estaque, 2012; Oteros-Rozas et al., 2013). As a result, many grasslands and wood pastures were also abandoned thus reinforcing rural depopulation and forest regrowth (Marull et al., 2010, 2014, 2015a, 2016a; Tello et al., 2014; Otero et al., 2015). The abandonment of rural and mountain areas were intensified by the European Common Agricultural Policy during the Second Globalization from 1986 onwards (Poyatos et al., 2003; Bielsa et al., 2005; Lasanta-Martínez et al., 2005 and 2006; Motet et al. 2006; Serra et al. 2008).

2.2 Cartographic sources and methods

To assess the LULCC in the study area, we have combined three digital maps drawn in 1868, 1956 and 2005. The oldest one corresponds to the Forest Map of Berga and Manresa District Councils (approximately equivalent to current Catalan *comarques* or counties) drawn at scale 1:200000 by the Commission for the Forestry Map of the Iberian Peninsula, created by the Royal Decree of the Spanish *Ministerio de Fomento* and published by the General Commission of Statistics of the Spanish Kingdom in 1868 (Casals, 2008). The map describes each dominant species in forestland units together with cropland patches devoted to grains and arboriculture. We have digitised and geo-referenced the image provided by the Nature Dataset of the Spanish Ministry of Agriculture, Food and Environment (www.magrama.gob.es/es/biodiversidad/servicios/banco-datos-naturaleza/). The land cover map of 1956 at scale 1:50000 comes from the digital photointerpretation of the black and white orthophoto map generated from the aerial photos taken by the U.S. Army Flight in 1956-57, provided by the Barcelona province Council and CREAF (www.sitxell.eu/en/mapa_historics.asp). The 2005 digital map corresponds to the third edition of the Land Cover Map of Catalonia, also generated by photointerpretation made in the CREAF from the colour orthophoto map provided by the Cartographic Institute of Catalonia at scale 1:5000 with more than 100 land cover and land use categories (www.creaf.uab.cat).

Maps of 1868 and 1956 were ortho-corrected using topographic maps and digital elevation models to be fitted to current projection and datum (UTM 31N, ED50), and to reduce their spatial inaccuracies. Because of the different data sources and methods used for their production, these maps initially showed contrasting spatial and thematic resolution making comparisons difficult. For this reason, their specific land use and cover labels were classified into six categories: Forest (U1), Herbaceous crops (U2), Woody crops (U3), Riverine habitats (U4), Water (U5), Scrublands and grasslands (U6). To these, two new categories were incorporated: Main roads (U7) and Urban areas (U8). Also, they were generalized to a minimum area of 10 ha for agricultural and forestry uses (U1, U2, U3 and U6), and 1000 m^2 for the rest (Fig 1), as corresponded to the minimum mapping areas for these categories in the least detailed map (i.e. that of 1868).

The historical land-use changes affecting forests have been assessed through Forest Land-Use Change Maps (FLUCM), obtained for two time periods (1868-1956 and 1956-2005) by combining the corresponding land cover maps. Four forest dynamic categories were considered in these FLUCM: Forest Maintenance (*FM*) when forestland uses U1 were kept along the corresponding time; Deforestation Processes (*DP*) when the U1 category became U2, U3 or U6; Reforestation Process (*RP*) when the reverse process occurred. The rest of land-use changes have been grouped into Other Processes (OP), which include maintenance and transitions within non-forest uses (cropland, pastureland, and non-vegetated categories like rocky areas, reservoirs, quarries and open pit mines, roads, highways, urban areas and industrial sites). All map reclassification, generalization, and combination tasks were performed using the Miramon GIS (www.creaf.uab.es/miramon)

2.3 Multi-regression analysis of the main factors behind new forestation in space and time

In order to assess the environmental and socioeconomic factors at stake, a set of proxies for forest accessibility and productivity has been selected: 1) Average annual radiation, indicative of energy available for forest growth; 2) Elevation, indicative of forest accessibility together with climate conditions (elevation is related to average temperature and rainfall) thus affecting again forest productivity; 3) Slope, indicative of forestland accessibility but also of forest productivity (e.g. through effects on soil depth); 4) Distance to main cities and towns in the region, indicative of travel costs to consumption places of forest products; and 5) Distance to villages, indicative of lumberjack's accessibility to forests.

Average annual radiation was obtained from the Climatic Digital Atlas of Catalonia, a set of raster climatic models obtained through statistical techniques (multiple regressions with residual correction) and spatial interpolation of data collected from meteorological stations in Catalonia (Ninyerola et al., 2000). The rest of variables were derived from official digital elevation models and topographic maps of Catalonia, in GIS environment.

A set of 3,000 points was randomly selected over the current forest area in the 2005 Land Cover Map. By using layer combination tools of the MiraMon GIS, these points were combined with the FLUCPM and the selected proxies for forest accessibility and productivity. Generalized Linear Models with binomial error were performed to ascertain the association of pre-existing (*FM*) versus new (*RP*) forestlands with the abovementioned environmental and socioeconomic factors, separately for the two studied periods (1868-1956 and 1956-2005). This allowed us to disentangle the effect of environmental factors of 'first nature' from other factors of 'second nature' (*sensu* Cronon, 1991; Krugman, 1993) such as human settlement and activity assessed through the forest distances to nearby villages or farther towns and cities.

2.4 Evidence from forest inventories and forestry statistics

The age of trees may also inform us about past disturbances that forestland has experienced. Two forest inventories exist for the study area: the Ecological Forest Inventory of Catalonia (IEFC, 1989-1998) by the Autonomous Government of Catalonia, and the Third National Forest Inventory (IFN3 2005) by the Spanish Ministry of Agriculture. By using their sampling areas (217 plots in the IEFC and 1,179 in the IFN3) we obtained data on the age of the trees in 1994, and on the structural forest parameters in 2005 (tree density in trees/ha, and diametric distribution patterns in cm). Plots with a tree cover lower than a basal area of $5m^2$/ha (equivalent to a forest crown coverage < 20%) have been removed, leaving a dataset of 972 dense woodland units for the analysis. Dominant tree species can be compared in these sample plots with the ones in 1868 and 2005, but not in 1956 due to the limitations of aerial photointerpretation.

In order to ascertain that the main driver behind these woodland patterns in forest cover, species distribution and age composition is anthropogenic, historical forestry data is needed. Unfortunately, official data of timber and firewood harvested encompassing all types of private, public and communal woodland only started to be recorded in Spain in the 1940s and the series were incomplete from 1940 to 1945. We have taken the data from the Instituto Nacional de Estadística (www.ine.es/en/welcome.shtml), available at provincial level for the period 1940-1971, and afterwards from the Institut d'Estadística de Catalunya (www.idescat.cat/en/).

3. Results

3.1. Analysis of forest transition from cartographic sources

FT is a process driven by both local and global factors, which results in regional and local changes in forest cover (Barbier et al., 2010; Rudel et al., 2010; Meyfroidt and Lambin, 2011). Comparing the three sequential maps and the categories it can be inferred that in the study area forest land cover increased from 49.1% in 1868 to 52.7% in 1956, and then 58.3% in 2005 (Fig. 2 and Fig 3).

Figure 2. Land-Use Land Cover maps of the study area in 1868, 1956 and 2005.

Figure 3. Land-Use and Land Cover categories of the study area in 1868, 1956 and 2005.

Throughout the whole study period (1868-2005), land cover changes affecting forests have been different during these two stages (Fig 4).

Figure 4. Forest Land-Use Change Process Map (FLUCPM) of the study area from 1868 to 1956 (left map) and 1956 to 2005 (right map).

From 1868 to 1956 forests were only maintained in 29.2% of the area (*FM*), whereas 23.6% was affected by reforestation (*RP*), 19.7% by deforestation (*DP*) mainly as a result of cropland expansion, and 27.4% experienced *OP*. In 1956-2005 forests were kept in 39.8% of the area (*FM*), 18.3% was reforested (*RP*), only 12.5% was deforested (*DP*) and 29.4% underwent other changes (*OP*) which include cropland and pastureland maintenance together with growing built-up land (Fig 5).

Figure 5. Proportional areal distributions of the four categories of dynamic land-use change in 1868-1956 and 1956-2005 according to the FLUCPM of the case study (FM: Forest Maintenance; RP: Reforestation process; DP: Deforestation Process; OP: Other Processes).

Subtracting *DP* to *RP* we get a forest net gain in 4% of the territory in 1868-1956, and 5.8% from 1956 to 2005, which added to *FM* means 33.2 % of the total area in the former period and 45.6 % in the later period.

3.2. Statistical results of socio-ecological drivers behind forestland change

According to our statistical results, those forestlands that lasted longer from 1868 to 1956 were significantly associated to slope, distance to villages and distance to main cities. Slope and distance to cities were positively associated whereas distance to villages was negatively associated to new forests. Afterwards, from 1956 to 2005 reforestation was significantly associated to slope, elevation and annual radiation. Annual radiation and elevation were positively associated, and slope was negatively associated to new forests (Table 1). The rest of factors were not significantly associated to any forest type.

Table 1. Generalized Linear Models with binomial error source exploring the association of new and pre-existing forest with the selected socio-ecological factors for the time periods studied.

Table 2 shows that local population increase was negatively associated to mean elevation from 1956 to 2005, indicating that mountain villages showed the earlier and highest depopulation. This association also appears during the 1868-1956 time period, but it was only marginally significant.

Table 2. General linear models exploring the association of population increase in municipalities with the selected environmental factors for the studied time periods.

Hence, there has been a shift in the environmental patterns of forest recovery across time. Formerly (from 150 to 60 years ago) new forests appeared in steep, less accessible and less productive areas closer to villages but farther from main cities than pre-existing forests. In contrast, new forests in the 1956-2005 period developed in higher, less steep lands that received higher radiation than pre-existing forests (Fig 6):

Figure 6. Plot effects of significant environmental correlates (x-axis) on the probability that forest is new (y-axis), obtained from the performed binomial models (black line, mean effect; dotted red lines, 95% confidence intervals). Effects for the 1868-1956 and 1956-2005 periods are shown, respectively, on the left and the right columns.

3.3. Evidence of forest regeneration according to forest inventories

The dataset provided by the Ecological Forest Inventory of Catalonia of 1994 allowed us to establish the average period of forest regeneration in a sample of plots across the study area—although with a very low representation of holm oak and oak woods. This data on ages and structure of trees underlines once more the turning point of the 1940s and 1950s. We can see in Fig 7 that 24% of plots had trees born between 1941 and 1950, whereas 47% of the plots include trees born between 1941 and 1960, and 23% from 1961 to 1980. It is remarkable that only 2% of the plots have trees born in the 19th century and 7% in 1901-1920, which highlights the overpressure exerted in forestland up to the 1950s.

Figure 7 Percentages of forest plots of different tree species sampled in the study area that started to regenerate in different periods from 1861 to 1980, according to the Ecological Forest Inventory of Catalonia (IEFC, 1989-1998).

If these values are compared selecting the plots corresponding to pure stands of the most common species, we found that for *Pinus halepensis* (Ph), *Pinus nigra* (Pn) and

Quercus spp. (Qsp) the majority of trees were born from 1941 to 1960, while *Pinus sylvestris* (Ps) was regenerated between 1951 and 1970. Oldest trees correspond to *Pinus sylvestris* and *Pinus nigra* species. This evidence on the ages and structure of trees allows us to check the historical data provided by the cadastral maps and statistics, and supplements the limits of their accuracy.

3.4 Historical series of forestry from the 1950s onwards

As explained, that a forest land unit was kept classified as woodland in the three maps of 1868, 1956 and 2005, does not say anything about its actual stand density index, the age distribution of trees, and species composition. Forest quality and maturity also depend on the exploitation exerted in the meantime, which may range from overuse and deforestation to underuse and regrowth. The historical series of forestry harvest is shown in Fig 8.

Figure 8. Timber and firewood harvested from woodland in the province of Barcelona, 1946-2005.

The above historical series of forestry has to be assessed taking into account levels as well as trends. By looking at levels, the most apparent feature is the pervasive forest abandonment from the 1950s onwards given that the average forestry harvest per year only represents around one fifth of the Net Primary Production of woodland (Cervera et al., 2015), and it is highly concentrated on the most productive and accessible areas. Trends are more complex to understand. First of all, we deem that the increase registered from 1946 to 1950 was more of the permits transacted, than the actual wood removal, with an annual average 60% lower than in the following period up to the seventies (Grupo de Estudios de Historia Rural, 2003). Later on, logging and firewood harvested stood stagnant at the abovementioned low levels for three decades.

4. Discussion

4.1. Increasingly 'spasmodic' forestry in Mediterranean forests

Forestry permits from the 1980s onwards showed decreasing logging (Fig 8) followed by an upward trend, whereas firewood harvesting decreased, and while both series became more volatile (the coefficients of variation of forestry removals increase from 1970 onwards, and they have a positive correlation of 0.40 with time). We suspect that this recent rise in timber harvest might have been an unintended result of big fires and other seemingly 'natural' disturbances, like trees that fell owing to strong winds and snow. In order to explore this hypothesis, we need to differentiate the underlying trend of the mainstream forestry from the 'spasmodic forestry' undergone after each wildfire experienced by these large woodland areas kept underexploited or unused. Therefore, we have adjusted the series of forest biomass harvested by subtracting the amount removed during the three years following each wildfire (that is, fires with an extent greater than the series' average of 2,500 ha) from woodlands burnt throughout the period 1975-2005. This way we obtain something like a trend of a baseline stream flow of forestry harvest not related to wildfires, once the 'spasmodic' removals after wildfires have been left aside (Fig 9).

Figure 9. Assessing trends of forest biomass harvested from woodland in the province of Barcelona, before and after taking into account the effects of forest fires, 1975-2005.

Along these 30 years there have been four main wildfire periods in the province of Barcelona. An area of 42,175 ha was burnt in 8 years from 1978 to 1988, and 12% of all harvest might be attributed to these fires. From 1989 to 1997 a total of 45,350 ha were burnt in only 4 years, including the great fire of 1994, and withdrawals stemming from fires grew up to 39% of the total. Then 12,772 ha were burnt in 3 years from 1998 to 2000, and the corresponding 'spasmodic' harvests might have weight 20% of the total. Finally, from 2003 to 2005 some 8,151 ha were burnt in 3 years, with 18% of possible post-fire removals. Throughout the period, forest fires might have led to an average timber removal of some 12 m^3 per burnt ha—a moderate amount which only includes the still profitable logs and excludes other useless biomass triturated or left abandoned in the burnt forests, while areas affected by wildfires also include some forestlands of very low tree density (with FCC up to 5%). Hence, considering the different tree coverage and compared to the average current harvest of timber, 12 m^3/ha is a relevant figure. According to this, forest abandonment might have been far more widespread than what is seen in official logging figures that do not distinguish normal forestry management from removals of burnt logs after wildfires.

4.2. Framing the long-term dynamics experienced by current woodlands: From overuse to abandonment

In this section we are going to interpret from an environmental history standpoint all the data provided so far on forest change in the study area from 1868 to 2005, using as a backdrop the existing literature that links LULCC with the long-term socio-ecological transitions (Fischer-Kowalski and Haberl, 2007; Marull et al., 2010; González de Molina and Toledo, 2014). By combining all the methods and results explained we can infer a complex process of FT that came along in two markedly different stages, driven by the main socio-ecological factors identified. The first stage lasted until the 1950s and entailed strong pressure over the remaining forestland threatened with deforestation, even though a number of small mountain villages started to be depopulated already in the mid-19th century leading to local reforestations in some highest and steepest areas (Figs 1, 4 and 7). Hence, reforestation began in highland, remote areas that were depopulated and abandoned (Basnou et al., 2013) while, at the same time, cropland was being expanded at the expense of forestland in lowland areas—mainly vineyards up to the grape *Phylloxera* Plague in the 1880s and 1890s (Badia-Miró et al., 2010). This may explain why deforestation was still at stake up to 1956 (Figs 3, 4, 6 and Table 1), while reforestation only took place in steeper areas closer to some higher mountain villages and far away from the road network (Marcantonio et al., 2013).

Then the faster and wider stage of FT began in the 1960s, when the complete industrialization of farming concentrated in flatter soils easier to mechanize and irrigate took place, whereas steeper areas became economically marginal, depopulated and abandoned. Rural abandonment largely explains why a new stage of FT ensued characterized by a vast reforestation of former cropland areas in the southern lowlands, and of former pasturelands in the Pyrenean highlands of the study area. Forest encroachment accelerated, and extended also in some flatter and more productive soils (Fig 5 and Table 2) due to their lost profitability for agricultural uses, while pasturelands shrunk after the cessation of extensive livestock grazing and transhumance between the Pyrenees and the lowlands (Fig 2). This LULCC has had a deep socio-ecological impact. After three or four decades of forestland and pastureland abandonment, following nearly two centuries of an increasing human pressure over more and more deforested woodland, the prevailing fire regime has deeply changed in the study area as in many other Mediterranean regions (Lloret et al., 2002). Wildfires have played a major role due to the unavoidable removal of burnt logs after each fire, through what we have considered a 'spasmodic forestry' (Fig 9). These new types of disturbances have a lot to do with the long-term swing from overuse to abandonment suffered by these woods which grew along the FT.

Behind these changes experienced by woodland cover in the study area underlies a relevant socio-ecological transition (Fischer-Kowalski and Haberl, 2007; Marull et al., 2010; González de Molina and Toledo, 2014). Overuse was triggered by a traditional agroforestry economy, still largely based on the indirect capture of solar energy through biomass flows, which entailed a high 'land cost' to provide for the biotic components of human consumption baskets (Guzmán and González de Molina, 2009). Underuse and abandonment have been carried out by the current oil-based economy that has ended with the former multipurpose integrated management of cropland, woodland, pastures and livestock husbandry, and has transferred the former land cost to foreign countries where Catalan imports come from (Marull et al., 2010; Agnoletti, 2014). After Spain joined the EU in 1986, wood imports soared while rural abandonment and the scope of unmanaged forests increased (Infante-Amate et al., 2015; Soto et al., 2016). During these years the network of natural gas pipes reached many inward towns and villages, new oil facilities for heating scattered houses were built, whereas globalization allowed for growing imports of biomass pellets from abroad. The steep decrease registered in the stream-flow of 'normal' (not 'spasmodic') firewood harvest, and the downward trend with a slower pace found when we look at the evolution of total forest biomass harvested by adding up logging and firewood, clearly fits with that (Fig 7).

4.3 Current state of forests and prospects for improvement

Low capitalization is a pervasive feature of these forests so prone to ‘natural’ disturbances like wildfires. In 1994 the woods of the study area were rather young, with an average age of 48 years (+/- 18.2 years). Tree ages were quite similar in different areas, regardless of the forest LULCC carried out in each plot. The forest land of 1994 that so was in 1956, and remained forested or in a reforestation process since 1868, had an average age of 49 years (+/- 18-21 years)—exactly the same average age of trees in land units which remained permanent forests from 1956 to 2005, and only slightly higher than the 42 years (+/- 13 years) found in new forestland that grew from 1956 onwards. Moreover, the data on forest structure found in 2005 show that the lower diametric classes (10-15 cm CD) represent 58%, the middle ones (20-30 cm CD) amount to 41%, while the plots having trees with larger diameters (CD 35-40 cm) do not exceed 1% of the total.

By comparing the cover extent of major forest formations in current forests per historical type, we see that woods of Holm oak (*Quercus ilex*) and Oak (*Quercus faginea*, *Quercus pubescens*) accounted for 31% of tree cover in forests already existing in 1868, while the newest forests that appeared after 1956 only have 17% of fagaceae (Guirado et al., 2008). We can conclude that deforestation driven by farmland expansion had adversely affected the evolution of holm oaks. 21st century forests present 17% more of conifers, mainly due to an increase of Scots pine (*Pinus sylvestris*) and White pine (*Pinus halepensis*) which offset the 43% decrease of Black pine (*Pinus nigra*). Indeed, the woods most affected by anthropogenic uses have been the ones covered with *Pinus nigra* mainly due to the comparative advantage of *Pinus halepensis* regeneration after fires (Retana et al., 2002). The area of *Pinus nigra* was largely affected by a large wildfire in 1994 and decreased by 77% between 1993 and 2009 (http://blog.creaf.cat; accessed 31/07/2014).

Their current state reveals that these new Mediterranean woodlands are very homogeneous in terms of age composition and species diversity (Boada, 2003). They exhibit a low resilience to natural and anthropogenic disturbances, such as fires and tree falls owing to strong winds, snow and droughts—a worrying feature in a time of climate change. Even more, our results suggest that these regrowth forests seem to be trapped into some vicious circles. For example, the lack of forest regeneration after fires may lead to an increasing recurrence of wildfires creating a spiral of decline that might end up endangering woodland permanence (Eugenio et al., 2004; Díaz-Delgado et al., 2002). Low landscape heterogeneity makes them prone to wildfires. Although fire disturbances might lead to a temporary increase of land cover diversity, this opportunity is lost without an adequate forest management (Lloret et al., 2002).

Knowing the long-term disturbances these forests have experienced, from a former overuse and deforestation to the current forest encroachment of abandoned cropland and pastures, helps to understand why this FT has led to ecological degrading side effects. Overcoming them, and the ensuing ‘spasmodic forestry’, means improving their ecological state by means of an active ecological restoration through well-designed site-specific forestry, agricultural and livestock grazing practices (Honnay et al., 2004; Parrotta et al., 2006; Jackson and Hobbs, 2009; Rey Benayas et al., 2005, 2008, 2009; Cuesta et al., 2010).

5. Conclusion

Our analysis has shown that forestland in the Berguedà and Bages counties (Catalonia) has experienced a long-term swing from overexploitation and deforestation from 1868 up to 1956, towards a subsequent FT driven by underuse and abandonment. The spatial scope and temporality of this double process differed in different places, deeply affecting the current ecological state of each forestland unit, which can be better understood by taking into consideration the site-specific history they have experienced. While the former overuse, together with cropland expansion, not only shrunk forestland but reduced the quality of woods that remained up to 1956, the ensuing fast and widespread FT following rural exodus has entailed a woodland expansion without no quality recovery (Grove and Rackham, 2001; Boada, 2003; Marull et al., 2014; Otero et al., 2015). As a result, woods became less resilient and more vulnerable to climate change (Kröel-Dulay et al., 2015). Our environmental historical analysis suggests that regrowth forests are caught in a low ecological quality trap. Their low resilience prevents them from maturing, and their low ripeness keeps them in a low resilience state. Wildfires become an epitome of this: the lack of an adequate forest management is leading to a 'spasmodic forestry' driven by the emerging disturbance regimes, like increasing wildfires, which prevents overcoming their bad ecological state.

According to this diagnosis, the improvement of forests requires a reversal of their current abandonment. If society demands good forestlands able to provide wood and good supporting and regulating ecosystem services now and in the future, sustainable forest management needs to be promoted based on a better understanding of the historical processes that lay behind these woods. Knowing the diverging historical trends undergone by different woodland units, we can identify those areas that suffered deeper land-use changes, where the mosaic structure of cultural landscapes can be restored by recovering pastures and mixing them with cropland and forestland. This would entail restarting again the kind of intermediate disturbance that, according to the land-sharing approach would be good for biodiversity conservation (Svensson et al., 2012; Tscharntke et al., 2012a, 2012b). At the same time the restoration of landscape mosaics would be of help to make forests less vulnerable to wildfires. Those land units recently forested as a result of rural abandonment have to be differentiated from other areas that have remained forestland from a long time ago, and may keep better woodland soils. According to our data, they represent less than 30% of the area. These few plots of real mature forests with older trees and richer soils should be carefully located and preserved (Fig 7).

Only by treating differently each different woodland unit, and by recovering a multifunctional forest management integrated with farming and livestock husbandry, the ecological state of these forests can be improved making them more resilient to climate change, and able to provide supporting and regulating ecosystem services together with timber and firewood. The data assembled and the interpretation given in this article makes clear that for this task, history matters (Honnay et al., 2004; Parrotta et al., 2006; Gustavsson, et al., 2007; Jackson and Hobbs, 2009; Tello et al., 2006, 2014; Geri et al., 2010; Jones et al., 2011; Morán-Ordóñez et al., 2011; Plieninger, 2012; Gao and Liu, 2012; Echeverría et al., 2012; Basnou et al., 2016). Forest history can provide to private forestry managers and public land-use planners a good guidance of where to do one or another type of management, or none, within an integrated sustainable forestry approach. No doubt, given the limited spatial accuracy of the historical sources available, more fieldwork inventories, soil tests and high-resolution land-use change assessment of specific areas are needed (Geri et al., 2011; Lieske and Gribb, 2012; Fuchs et al., 2013, 2015). Yet the general overview provided by the socio-ecological history of these woods greatly facilitates the preliminary design of areas where a closer scrutiny would be required.

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Table 1

	Estimate	Std. Error	z value	Pr(>|z|)
1868-1956 period				
(Intercept)	-1,25E+03	5,76E+02	-2,175	0,02960 *
Annual radiation	6,71E-01	3,79E-01	1,772	0,07640 .
Elevation	4,12E-02	1,63E-01	0,253	0,80041
Slope	1,89E+01	6,29E+00	3,004	0,00266 **
Distance to main cities	3,61E-02	1,22E-02	2,973	0,00295 **
Distance to villages	-1,10E-01	1,57E-02	-7,031	2,05e-12 ***
1956-2005 period				
(Intercept)	-5,03E+03	6,26E+02	-8,038	9,12E-16 ***
Annual radiation	3,34E+00	4,26E-01	7,845	4,33E-15 ***
Elevation	6,31E-01	1,50E-01	4,200	2,67E-05 ***
Slope	-4,62E+01	6,63E+00	-6,968	3,22E-12 ***
Distance to main cities	-1,22E-02	1,11E-02	-1,099	0,272
Distance to villages	1,90E-03	1,48E-02	0,128	0,898

Source: our own, from the datasets explained in the text. · level of significance at 10%, *= level of significance at 5%, **=level of significance at 1%, ***= level of significance at 0,1%.

Table 2

1868-1956 period	Estimate	Std. Error	t-value	Pr(>|z|)
(Intercept)	-134,56	41,7	3,230	0,002 **
Distance to main cities	-0,003258	0,003637	-0,896	0,373
Elevation	-0,082336	0,043584	-1,889	0,063 ·
1956-2005 period	**Estimate**	**Std. Error**	**t-value**	**Pr(>|z|)**
(Intercept)	-104,4	39,9	-2,620	0,011*
Distance to main cities	-9,479E-04	3,945E-03	-0,271	0,786
Elevation	1,128E-01	4,190E-02	-2,691	0,009 **

Source: our own, from the datasets explained in the text. · level of significance at 10%, *= level of significance at 5%, **=level of significance at 1%.

Figure 1

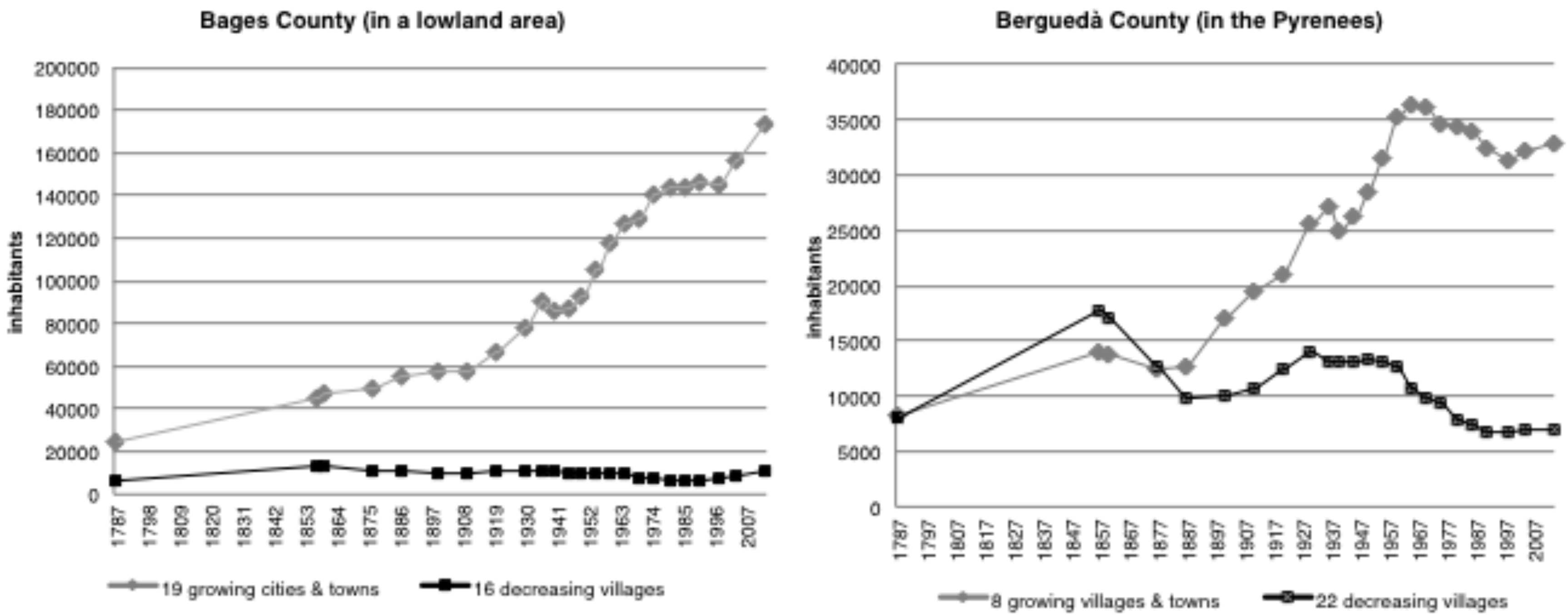

Bages County (in a lowland area)
inhabitants
19 growing cities & towns
16 decreasing villages
Berguedà County (in the Pyrenees)
inhabitants
8 growing villages & towns
22 decreasing villages

Figure 2

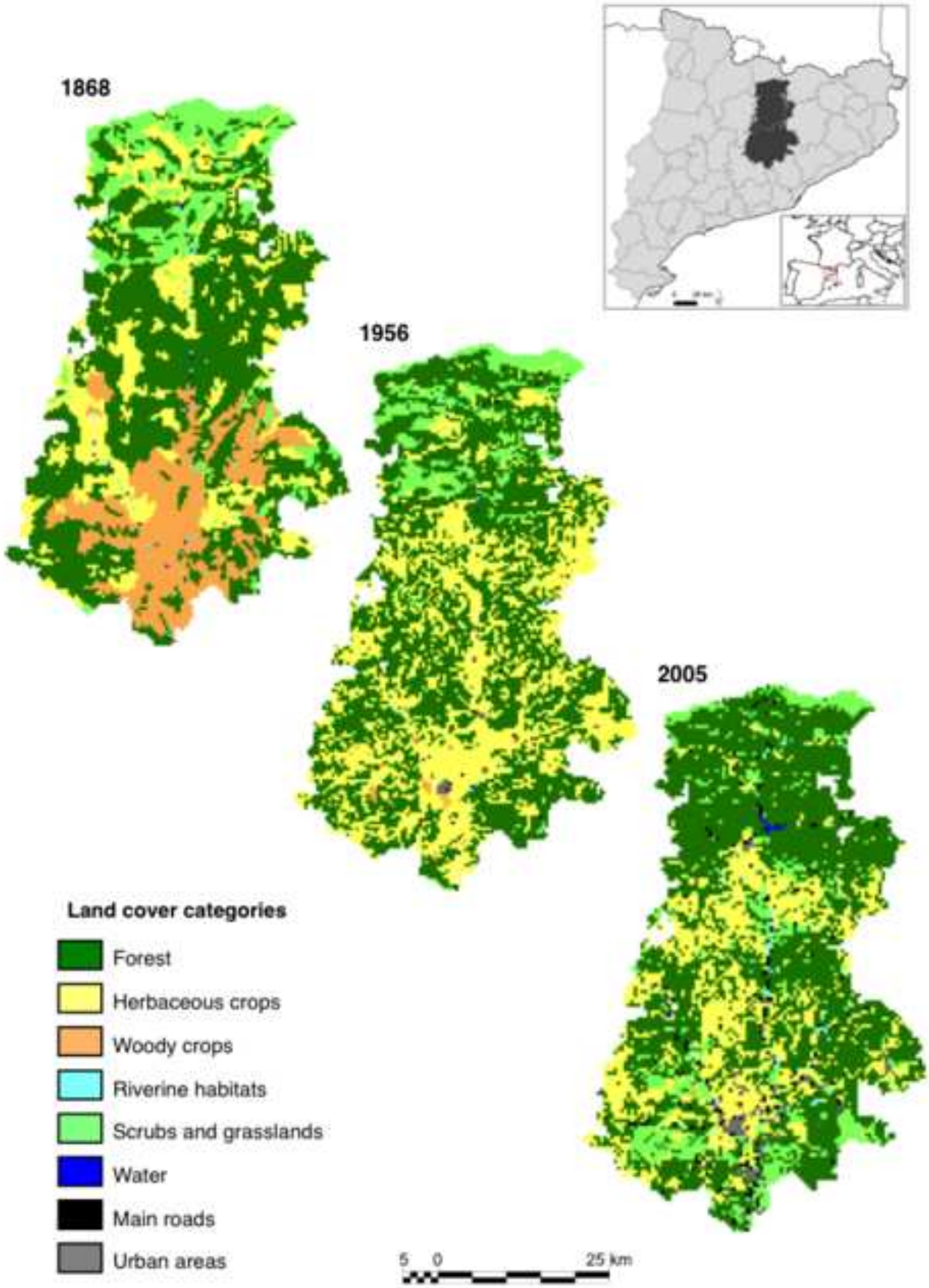

1868
1956
2005
Land cover categories
Forest
Herbaceous crops
Woody crops
Riverine habitats
Scrubs and grasslands
Water
Main roads
Urban areas
5
0
25 km

Figure 3

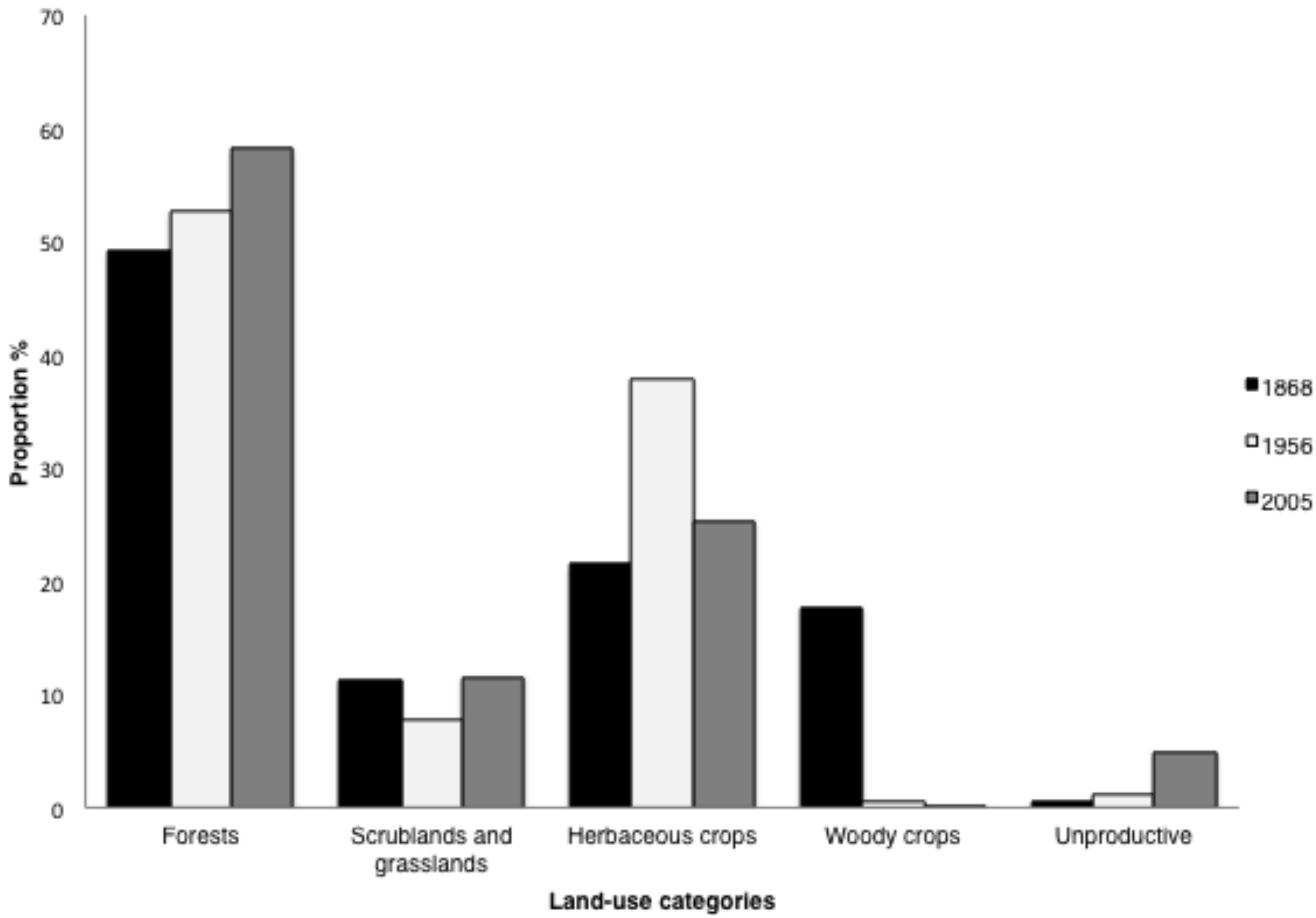

70
60
50
40
30
20
10
0
Proportion %
Forests
Scrublands and grasslands
Herbaceous crops
Woody crops
Unproductive
Land-use categories
1868
1956
2005

Figure 4

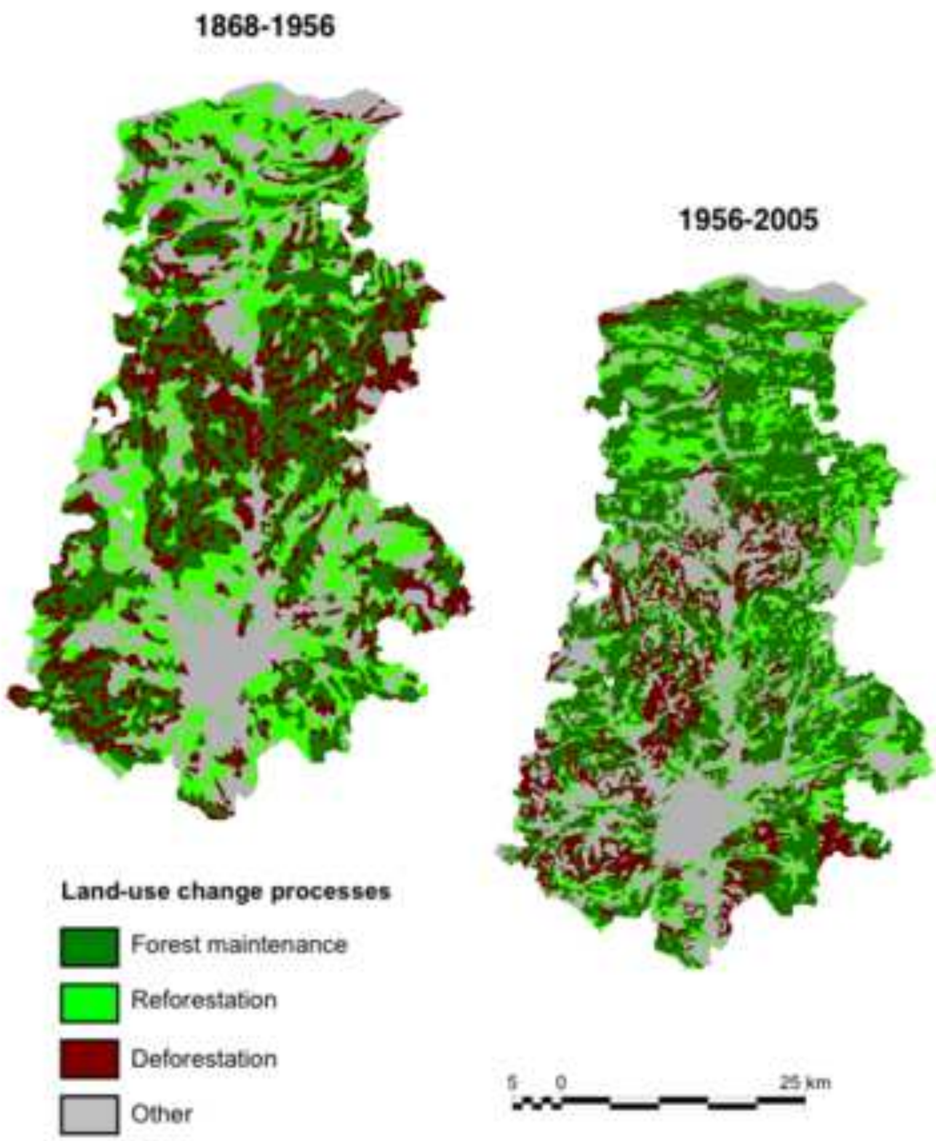

1868-1956
1956-2005
Land-use change processes
Forest maintenance
Reforestation
Deforestation
Other
5 0 25 km

Figure 5

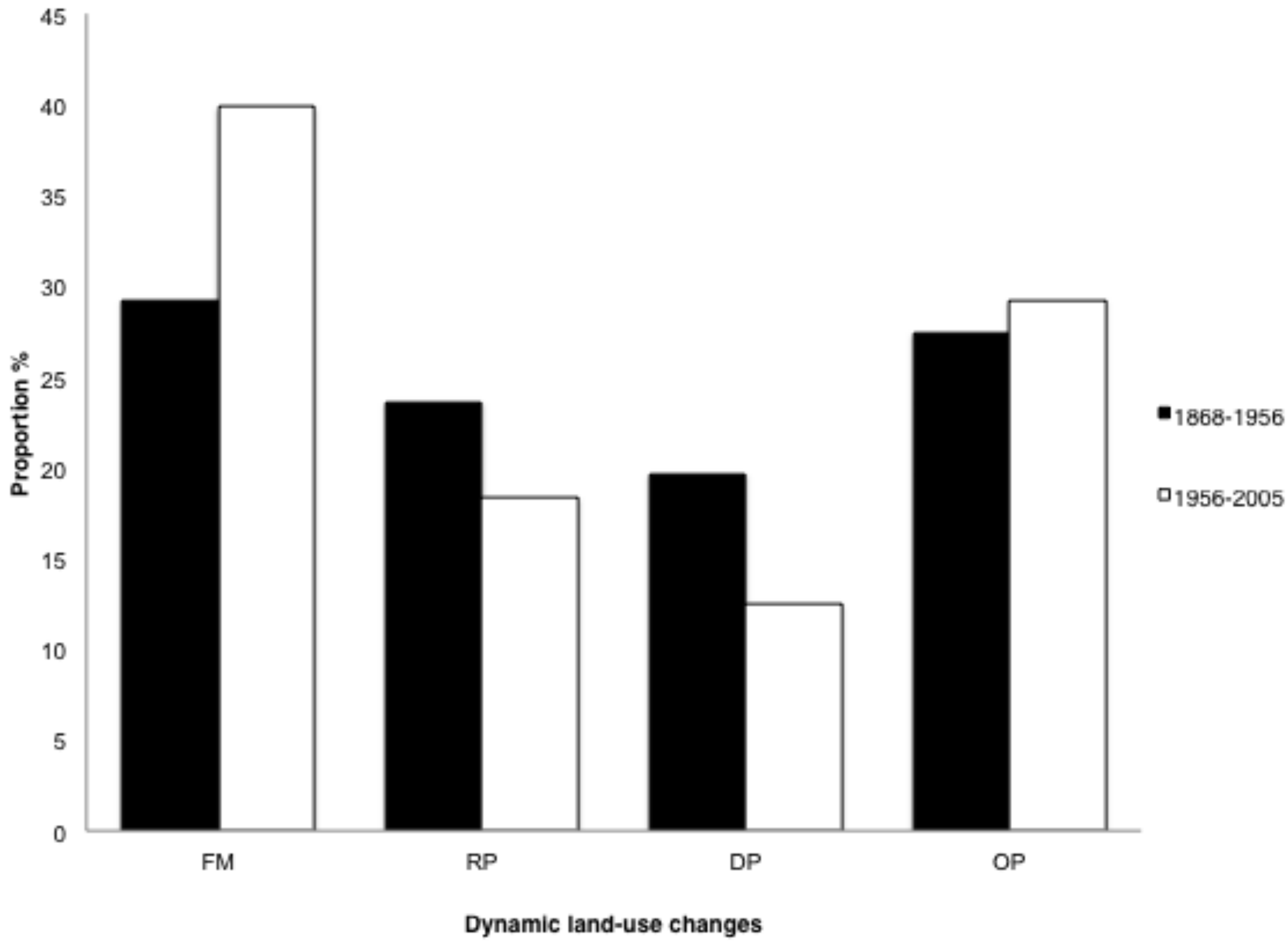

45
40
35
30
25
20
15
10
5
0
Proportion %
FM
RP
DP
OP
Dynamic land-use changes
1868-1956
1956-2005

Figure 6

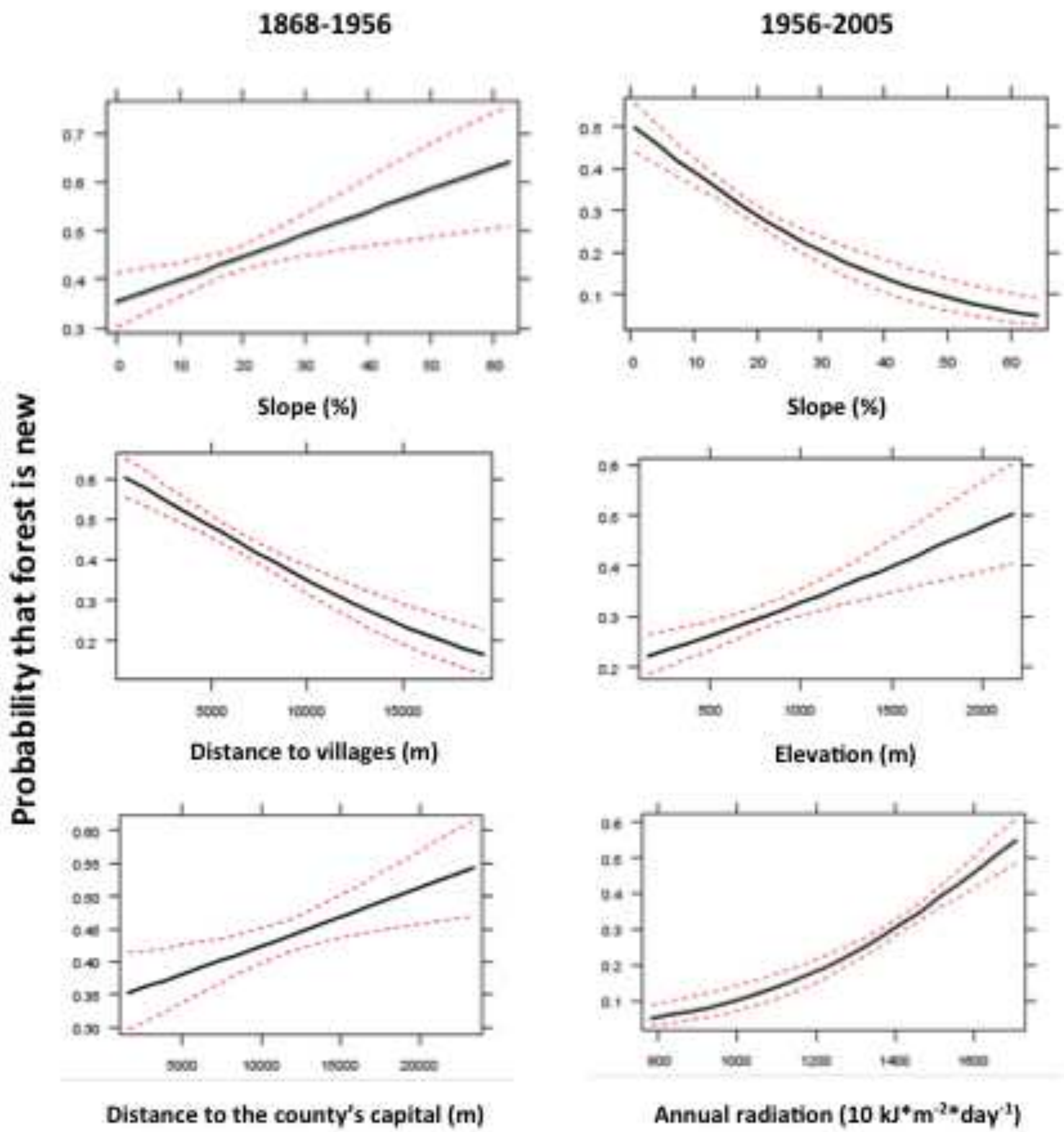
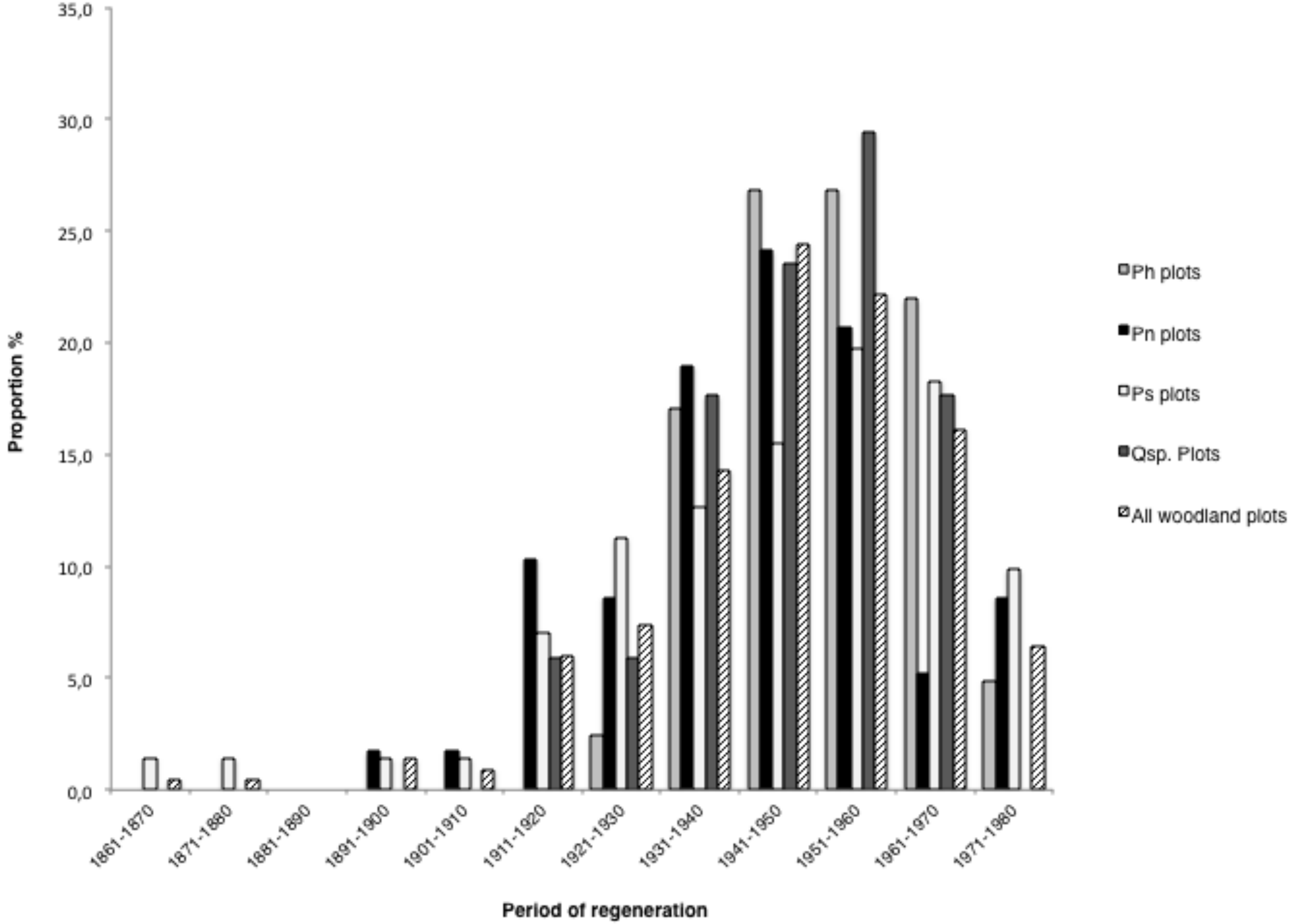

1868-1956
1956-2005
Probability that forest is new
Slope (%)
Slope (%)
Distance to villages (m)
Elevation (m)
Distance to the county's capital (m)
Annual radiation (10 kJ*m⁻²*day⁻¹)

Figure 7



35,0
30,0
25,0
20,0
15,0
10,0
5,0
0,0
Proportion %
1861-1870
1871-1880
1881-1890
1891-1900
1901-1910
1911-1920
1921-1930
1931-1940
1941-1950
1951-1960
1961-1970
1971-1980
Period of regeneration
Ph plots
Pn plots
Ps plots
Qsp. Plots
All woodland plots

Figure 8

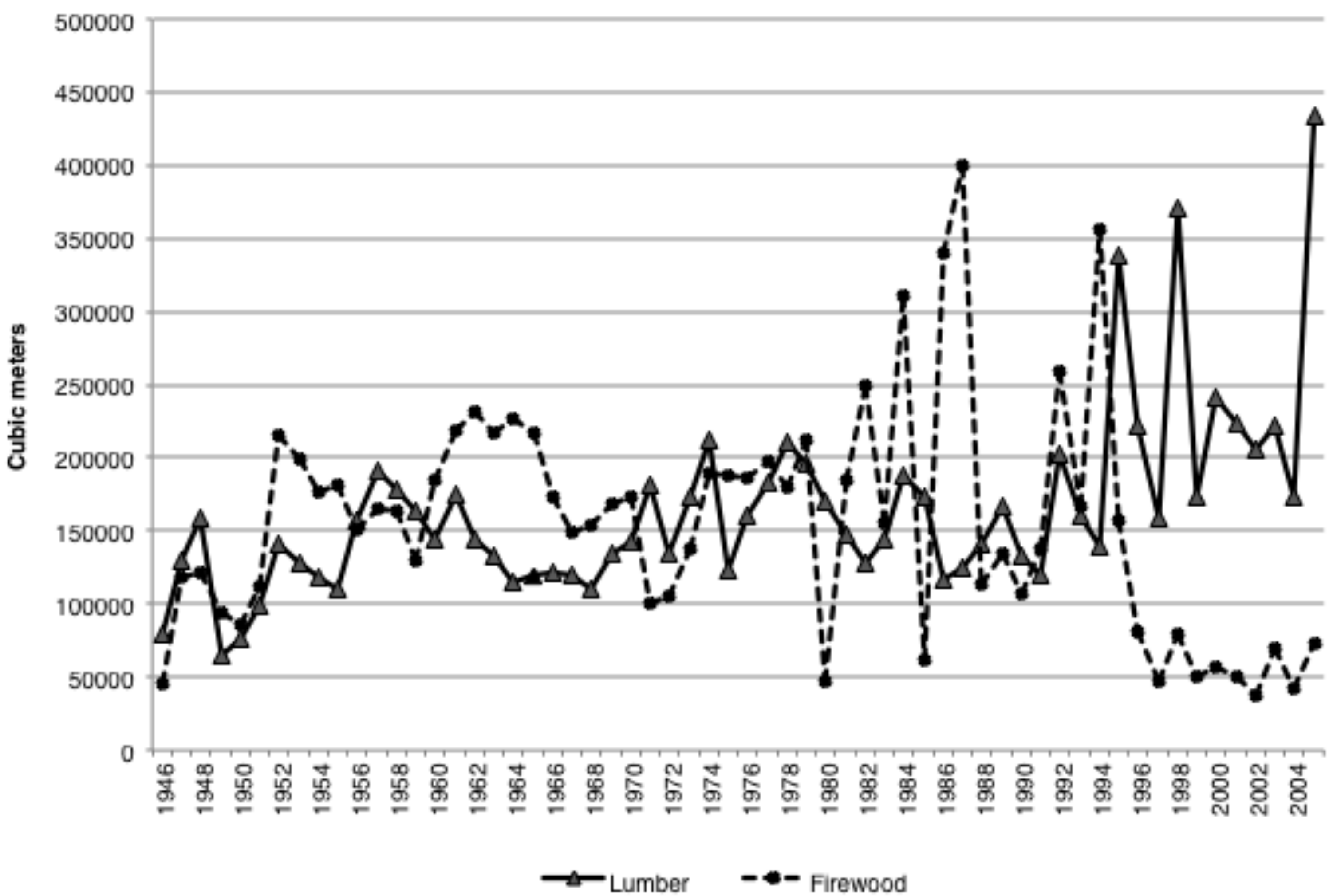
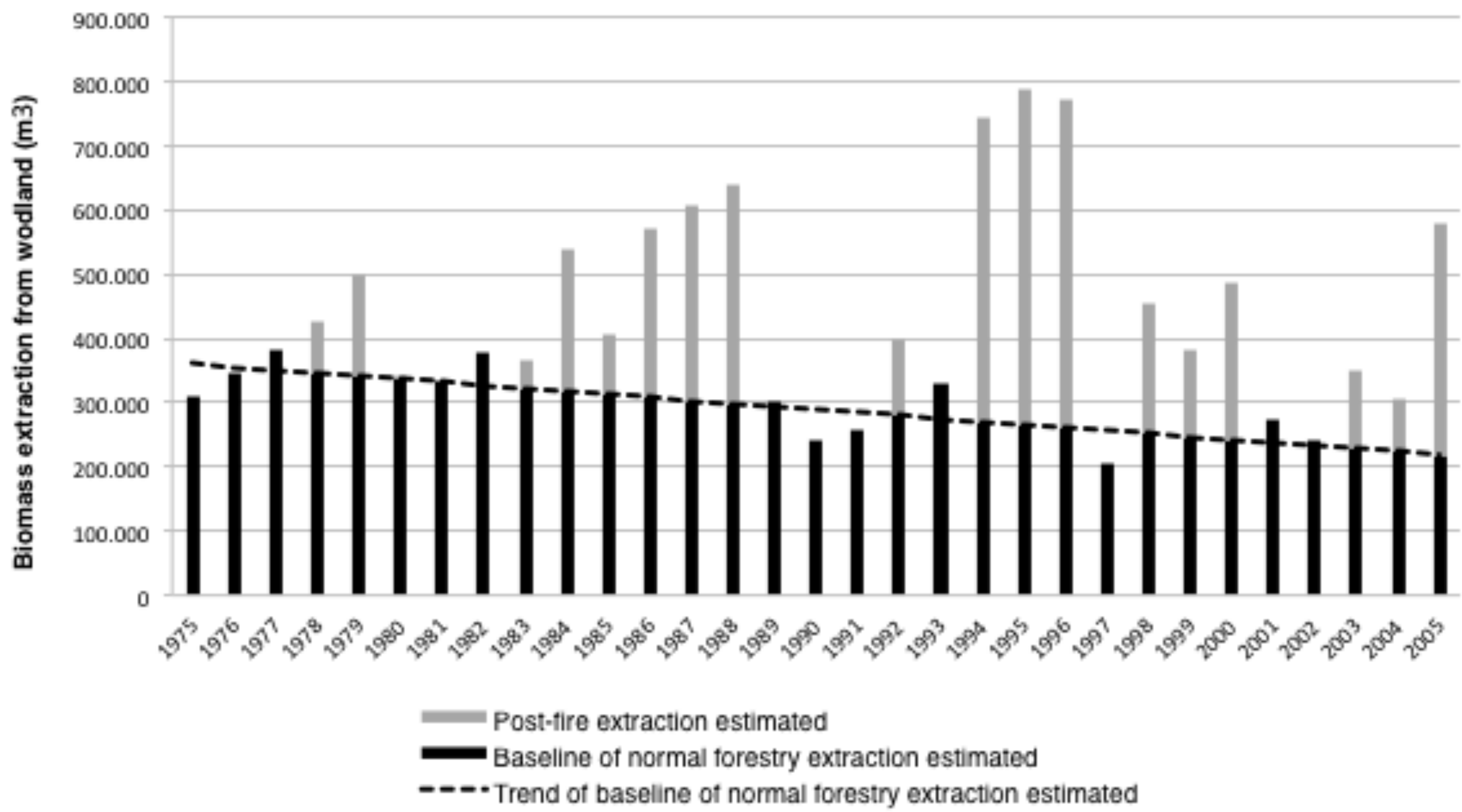

Cubic meters
500000
450000
400000
350000
300000
250000
200000
150000
100000
50000
0
1946
1948
1950
1952
1954
1956
1958
1960
1962
1964
1966
1968
1970
1972
1974
1976
1978
1980
1982
1984
1986
1988
1990
1992
1994
1996
1998
2000
2002
2004
Lumber
Firewood

Figure 9


Biomass extraction from wodland (m3)
900.000
800.000
700.000
600.000
500.000
400.000
300.000
200.000
100.000
0
1975
1976
1977
1978
1979
1980
1981
1982
1983
1984
1985
1986
1987
1988
1989
1990
1991
1992
1993
1994
1995
1996
1997
1998
1999
2000
2001
2002
2003
2004
2005
Post-fire extraction estimated
Baseline of normal forestry extraction estimated
Trend of baseline of normal forestry extraction estimated