Monitoring degradation in arid and semi-arid forests and woodlands: the case of the argan woodlands (Morocco)*

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Abstract

Arid and semi-arid forests and woodlands (hereafter called «dryland forests»), in spite of their ecological and social importance, have received little attention in land change studies. Growing evidence shows that these forests have been receding at very high rates in many places, suggesting a need for a better understanding of the processes and causes of dryland forest degradation. Changes in the extent of dryland forests are debated in part because estimates of forest and woodland areas in drylands are uncertain. Causal explanations of the degradation tend to draw on the literature on desertification and tropical deforestation, and to emphasize either local or remote, and either social or biophysical drivers. This study contributes to a better understanding of dryland forest degradation as a basis for conservation policies. Firstly, we argue that monitoring arid and semi-arid forests and woodlands using area estimates may lead to an underestimation of the severity of change because tree density change often exceeds area change. Secondly, we argue that the analysis of degradation processes in these multifunctional landscapes should integrate both local and remote, and both social and biophysical factors. We use a case study of degradation in the *argania* woodlands in semi-arid to arid Southwest Morocco to test these two claims. We used gridded tree counts on aerial photographs and satellite images to estimate forest change between 1970 and 2007, and we tested several possible causes of change on the basis of original socio-economic field surveys and climatic and topographic data. We found that forest density declined by 44.5% during this period, a figure that is significantly underestimated if forest area change is used as a measure of degradation. Increasing aridity and, to a lesser extent, fuelwood extraction were related to forest decline. No effect of grazing by local livestock was found.

Keywords: drylands; degradation; deforestation; Morocco; remote sensing

1. Introduction

Deforestation in tropical forests and land degradation in drylands have been studied extensively. However degradation processes in arid and semi-arid forests and woodlands (hereafter called «dryland forest») have received far less attention (Grainger, 1999). Dryland forests account for some 6% of the world's forests (Malagnoux, 2007). They include most of North African and Sahelian forests and woodlands, parts of the Miombo woodlands and bushlands in Central and Southern Africa, the Dry Chaco, Caatinga and Mediterranean ecosystems in South America, the Brigalow Belt in Australia, as well as parts of Mexico, the United States and Central and South Asia. They provide key ecosystem goods and services to over one billion people living within the arid and semi-arid climatic zones (Millenium Ecosystem Assessment, 2005). Ecosystem goods provided by dryland forests include fodder, fuelwood, wood for tool-making and other uses, medicines, herbs and other non-timber forest products and tradable goods; ecosystem services include soil stabilization, climate change mitigation, water conservation, and erosion and desertification control (Malagnoux, 2007; FAO, 2010; Rotenberg & Yakir, 2010). Several authors demonstrated the importance of forest products for local populations, especially for the rural poor (Sunderlin et al., 2005; Shackleton, 2007; Salehi et al., 2010). Dryland forests have been receding in several areas, sometimes at rates comparable to those found in tropical rainforests (Zak et al., 2004; Seabrook, 2006). In spite of all this, still surprisingly little is known today as to the extent and processes of degradation in these forests.

In the Sudano-Sahelian region, the question of whether there is a secular trend in forest degradation at the regional-scale is debated (see for example Schlesinger, 1996; Chamard & Courel, 1999; Gonzales, 2001). In North Africa there is wide agreement that forests have been receding (Benabid, 1996; Sutton & Zaimeche, 1996; Benbrahim et al., 2004) but little scientific evidence on the extent of the degradation has been produced. Degradation and deforestation in South American drylands have been better documented. Several figures of deforestation have been published, from a 29% decrease between 1972 and 2007 in the central Chaco (Gasparri & Grau, 2009), to over 80% decrease (including conversion of primary to secondary forests) between 1969 and 1999 at the southern edge of the Arid Chaco (Zak et al., 2004). In central Chile, about 42% of the original dryland forest area was lost between 1975 and 2008 (Schulz 2010). In the Australian Brigalow Belt, the clearing of native woody vegetation has a long history, and was still going on in recent decades, with an estimated 1.10^6 ha of woody vegetation cleared between 1995 and 2004 (Seabrook et al., 2006). In other areas, such as Southern Africa and Central and South Asia, evidence has been scarce.

One reason for the absence of a clear picture of dryland forest change worldwide is the difficulty of monitoring change with classical tools such as national forest inventories and surface change estimates. Dryland forests are often not included in national forest inventories (Grainger, 1999). Surface change estimates vary widely and, where they are available and consistent, they may still not reflect actual biomass and productivity changes, which may be under- or over-estimated depending on whether density is increasing or decreasing (Rautiainen et al., 2011; Conchedda et al. 2011). We argue that surface estimates may lead to an underestimation of the severity of change in dryland forests because, in many of these forests, density is decreasing faster than their area. Indeed, unlike in dense tropical rainforests, the main modes of exploitation of dryland forests (i.e., fuelwood harvesting, grazing, agriculture) do not lead to a complete clearance but rather a thinning of the forest (Grainger 1999).

The causes of dryland forest change are likewise difficult to track down. Evidence from the science of dryland degradation shows that land degradation processes in drylands are complex and involve interacting social and biophysical drivers (Geist & Lambin, 2004, Reynolds et al. 2007). Most of the literature on dryland degradation however focuses on local causes. The literature on deforestation in tropical forests, by contrast, focuses mostly on social and distal causes. This focus on either social or biophysical, and either local or remote drivers is also found throughout the literature on dryland forest degradation. Actually, causal explanations of dryland forest degradation can be classified in four discourses, each emphasizing the remote or the local, the social or the biophysical. These are briefly reviewed hereafter.

The *neo-Malthusian*, discourse explains the degradation of dryland forests by the overuse of resources, usually as a consequence of population increase. Overuse can take the form of overgrazing, fuelwood harvesting, or trade in forest products such as fruits and seeds. Human-induced degradation may trigger self-sustaining degradation patterns, where aging and less resilient forests are more vulnerable to shocks, such as severe droughts and pest invasions (see Barbero et al., 1990; Buttoud, 1994; Benabid, 1996; M'Hirit et al., 1998; Chamard & Courel, 1999; Singh, 2003; Odihi 2003; Ozer, 2004; Salehi et al., 2008; Schultz et al., 2010; Säumel et al., 2011).

The *climatic* discourse argues that climate change, and in particular the increasing length and frequency of droughts, causes trees to die. Fuelwood harvesting is not seen as a threat because people harvest mostly dead wood. Projections of rising hydric deficit in most climate change scenarios are the main cause for concern (see Benjaminsen, 1993; Nagothu, 2001; Hiemstra-van der Horst & Hovorka, 2009; Fensham et al., 2009; Hiernaux et al., 2009).

The *economic* discourse sees dryland forest degradation as a consequence of urbanization and the expansion of agriculture and rangelands, driven by favorable economic and institutional conditions (see Zak et al., 2004; Seabrook, 2006; Boletta et al., 2006; Gasparri & Grau, 2010; Schulz 2010).

The *political ecology* discourse questions the existence of human-induced degradation in dryland forests, and argues that the idea of degradation is used to justify intrusive land-use and development policies and maintain neo-colonial power structures. Where it happens degradation is often considered as a result of unsustainable activities encouraged by neoliberal economics, such as intensive irrigated agriculture, or of the marginalization of poor populations through the appropriation of land by elites (see Bush, 1997; McCann, 1999; Ribot, 1999; Nagothu, 2001; Katz, 2002; Davis, 2006).

These four discourses may have more or less relevance depending on the geographic and social environments considered. In many cases their shortcomings are obvious, and there is a case for a more integrated understanding of the degradation processes. Indeed, dryland forests are highly complex, multifunctional landscapes. We argue that change in dryland forests transcends these categories and can only be understood by taking into account both local and remote, and both biophysical and social drivers.

The objective of this study was to use the case of the argan woodlands in Southwest Morocco to test the claims that: (1) surface change values underestimate the degradation of dryland forests; and (2) changes in dryland forests can only be understood by taking into account both local and remote, and both biophysical and social drivers. We first constructed an integrated conceptual model of degradation in the argan woodlands based on the literature. Forest change was then estimated by gridded tree counts on aerial photographs and high resolution satellite images. Thirdly, household surveys and key informant interviews conducted during fieldwork in 2009-2010, along with climatic and topographic information, were used to estimate the contribution of fuelwood collection, grazing, and aridity to forest degradation.

2. Background

2.1 The argan woodlands

The argan tree (*Argania spinosa*) is endemic to Southwestern Morocco, where it covered about 950,000 hectares in 2010 (Lefhaili, 2010). The argan woodlands are part of the *Mediterranean Acacia-Argania dry woodlands* ecoregion (World Wildlife Fund, 2001). The area was designated as a UNESCO «Man and the Biosphere Reserve» in 1998. Tree density varies from under 30 trees/ha to over 80 trees/ha. Argan trees occur in different sizes, from shrubs lower than 1 meter to trees over 5 meters high. They can be found at altitudes ranging from the sea level to 1500m, and in areas with rainfall levels from 150 to 400 mm/year. The tree's ability to survive arid conditions is in part due to its deep-reaching roots and its ability to drop leaves in case of severe drought (Msanda et al., 2007).

Argan trees have traditionally provided multiple ecosystem goods and services including the provision of fruits from which argan oil is produced, leaves and young shoots eaten by sheep, goats and camels, and wood for fiber and fuel (M'hirit et al., 1998). Argan oil has become famous in the last two decades for its cosmetic virtues as a moisturizer and anti-wrinkles treatment, and has been exported at prices up to several hundred dollars a liter to Europe, Japan and the United States (Lybbert, 2010). Goats and sheep eat the oil cake, a by-product of argan oil production. Their manure is used as a fertilizer for cereal crops growing in the shade of argan trees, and traditionally they functioned as nut collectors by eating fruits and defecating the nuts in the barns. Argan wood has also been used traditionally for roofing, tool making, cooking and baking. Finally, old argan trees have a spiritual value and are frequently named after local saints, and linked to sacred places and religious festivals (Simenel, 2004). People from the argan region have developed a particular management system to protect fruits during maturation (Bourbouze & El Aich, 2005).

2.2 Forest changes

Reports on forest decline in the argan zone are not new, but evidence for it has been scarce. Rocher (1926), Boudy (1958) and Monnier (1965) reported widespread forest clearing in the first part of the 20th century, mostly attributed to fuelwood demand from the rapidly growing coastal cities and from Europe during the two world wars. McGregor et al. (2009) found a decrease in argan pollen concentration in sediments close to the Atlantic coast since the late 18th century, after a strong increase since 1700 AD. Forest officials reported a 1.16% (11100 ha) decrease in the extent of the argan forest between 1990 and 2005, or 740 ha/year, but provided no information as to how this was estimated (Lefhaili, 2010). El Yousfi (1988, cited in M'Hirit et al., 1998), comparing aerial photographs, showed that middle- and low-density forests were opened to cultivation on 45% of a 22000 ha study area in the Sous plain between 1969 and 1986. A recent report however found that only about 9% of a wider study area of 55000 ha, including the former 22000 ha, were cleared for cultivation between 1969 and 2006 (TTOBA, 2007), suggesting that not all the areas opened to cultivation were actually cleared of trees. This same report also found that 5% of the area was cleared for urbanization, and 32% of the area underwent some forest density decrease between these two dates.

2.3 Causes of change

Forest decline in the argan woodlands results from a combination of lack of regrowth and loss of trees, which are in turn controlled by biophysical and social factors (Figure 1). The lack, if not complete absence, of natural regrowth across the region has been acknowledged by many observers (M'Hirit et al., 1998; Tarrier & Benzyane, 2003; Nouaim, 2005; Bellefontaine 2010). It may have been caused partly by overgrazing, as animals eat new saplings (M'Hirit et al., 1998; Nouaim, 2005; Naggar, 2005; Aafi, 2007). Enclosure experiments in two areas (Culmsee, 2005; Taleb, 2007) found that regrowth does occur if grazing is excluded. Overgrazing may also damage soils, remove understory and cause erosion (M'Hirit et al., 1998; Aafi, 2007). Some authors also argue that excessive browsing by goats and camels damages the trees (M'Hirit et al., 1998; Nouaim, 2005; Aafi, 2007), whereas others suggest that moderate browsing may stimulate fruit production (Bourbouze & al., 2005). Seed harvesting for the production of argan oil may become unsustainable if no seeds are left on the ground to germinate, as is the case in some places (M'Hirit et al., 1998; Benghazi, 2007). Increasing prices for argan oil have probably led to an intensification of fruit collection since the 1990's (Lybbert, 2002).

Increasing aridity, reported at a regional scale by Esper et al. (2007), may also have adverse impacts on germination and seedling survival (Nouaim, 2005; see also Stour & Agoumi, 2009). Aridity was shown to increase developmental instability, which amounts to decreasing resilience (Alados & El Aich, 2008). Excessive hydric stress during long and severe droughts may increase mortality in tree stands, especially those on slopes and with high sun exposure (El Abidine, 2003). Increasing aridity can be due to climate variability or change, or to a change in micro-climatic conditions prompted by the clearing of trees and understory (M'Hirit et al., 1998; Nouaim 2005). Droughts may exacerbate grazing pressures on existing resources (M'Hirit et al., 1998).

Clearing of trees and understory also causes erosion that, along with soil degradation (in the form of salinization and loss of organic matter, mostly in irrigated areas), may render some soils unfavorable for germination (Bani Aameur & al., 2001; Nouaim, 2005; Aziki, 2008). Cities such as the Agadir-Ait Melloul conurbation, as well as smaller administrative centers, have been expanding into the argan forest (M'Hirit et al., 1998; Nouaim, 2005; TTOBA, 2007). Irrigated agriculture has likewise expanded into the forest, mostly in the Souss plain (El Yousfi, 1988; TTOBA, 2007). An indirect adverse effect of the expansion of irrigated agriculture is the lowering of the water table through over-pumping, which may increase vulnerability of trees to drought (M'Hirit et al., 1998; El Abidine, 2003; Nouaim, 2005; TTOBA, 2007). Finally, excessive fuelwood extraction, by the forest authorities and villagers, for trade and local consumption, has long been considered as an issue in the region (Monnier, 1965; M'Hirit et al., 1998; Bourbouze & El Aich, 2005; Nouaim, 2005; Faouzi, 2005; Aafi, 2007; Abourouh, 2007).

Many of the causal links depicted above are still partly hypothetical. Their relative importance is not known. Some might prove to be insignificant whereas some others may have been ignored. This model should thus be seen as a roadmap to the current understanding of the degradation of the argan forest rather than as a definite explanation of its causes. Figure 1 shows that both biophysical processes (increasing aridity, erosion) and social drivers (wood harvesting, overgrazing) may be involved. Several causes have both a local and a remote component – e.g., grazing by local or nomad herds, fuelwood consumption for local needs or for the market. While elements of the four discourses mentioned in the introduction can be found in the literature on argan woodlands, the neo-Malthusian discourse dominates, with most authors and decision-makers putting the blame on unsustainable firewood collection and overgrazing.

Our case study provides a measure of tree density and forest area change in a subset of the argan forest, and an estimation of the effect of three of the factors mentioned above, based on original field data: overgrazing, fuelwood extraction, and aridity. Concerning the other factors, either we could not collect biophysical data to analyze them (erosion, understory loss, lowering of the water table), or they were not detectable or irrelevant given the scale of the study. It is the case for the over-harvesting of seeds, which is a recent issue and whose effects will not be visible on aerial photographs. The expansion of irrigated agriculture and urbanization are nearly inexistent in the study area and occur mostly in places devoid of trees.

3. Material and methods

3.1 Study area

The study area is located in *Taroudant* province and covers 1100 km^2 or about 12% of the entire argan region (Figure 2). Fieldwork was conducted in five villages of that region (points A to E on Figure 1, total area 75 km^2). Argan trees are scattered (mostly under 30 trees/ha) and are almost the only tree species in the area, apart from a few acacias (*Acacia gummifera*) on slopes, and jujub bushes (*Ziziphus lotus*) on wad terraces. Argan trees occur in various morphologies, from low shrubs to tall trees. Their height rarely exceeds 5 to 6 meters, but their crown can reach diameters of over 10 meters. The bedrock consists mostly of dolomites, covered with quaternary silts in the plain. Soils can be deep in valleys but are thin or inexistent on slopes, where rocks and gravel dominate. Annual grasses cover most of the area after the rains, and some persist in the shade of argan trees. The area receives about 300 to 400 mm precipitation per year on average. Traditional land use is based on barley and wheat cultivation under argan trees, livestock herding (mostly goats in the hills, and sheeps in the plains), argan oil production, and wood and charcoal extraction for trade in times of drought. Since the 1980's, migration and remittances have become central elements of the economy.

3.2 Data

The detection of forest density change in El Faid rural district was performed on a series of 1970 black and white aerial photographs at 1/30000, and SPOT and Quickbird images from 2007 at respectively 2.5 and 1 m resolutions (color composites, both consulted in Google Earth^{tm}). The ASTER 25m resolution Global Digital Elevation Model (GDEM) was used in the geocorrection process and in the calculation of topography-related variables. Additionally, trees were counted for ground validation in 38 sample zones randomly located within the territories of the five villages where field surveys were conducted. Topographical features were used to delineate spatial units, resulting in zones of various sizes ranging from 0.8 to 23 ha (μ = 4.2 ha, stdv. = 4.78 ha). The larger zones correspond to low tree density areas. Temperature and precipitation data (1950 to 2005; 5° x 5° grid) come from the Climate Research Unit (University of East Anglia, UK)[1]. The Palmer

1 Data available at http://www.cru.uea.ac.uk/cru/data/

Drought Severity Index values (see the Methods section) are from Dai et al. (2004; 2.5° x 2.5° grid)[2].

Variables	Scale	Method	Source
Forest density change 1970-2007	District : 1 circle/km2 (n = 1067) Village : 8 circles/km2 (n = 622)	Visual counting in stratified random sample on aerial photographs and satellite images	Aerial photographs : 1/30.000 images from 1970 mission, *ANCFCC* Digiglobe (1m, 10/29/2007) and SPOT (2,5m ; 5/30/2007) (images (from Google Earth™)
Number of goats and sheep per village (1975 and 2009)	Village	Interviews of local customary authorities (n=6)	Field work 2009-2010
Contraventions for illegal wood extraction (number, location and origin of contravener)	Village	----	Data from Taroudant *Service Provincial des Eaux et Forêts* registers
Wood consumption (donkey loads per month)	Household	Household interviews (n=86)	Field work 2009-2010
Forest standing volume and natural increment	Whole argan region (aggregate values)	----	FAO FAR Country report, Morocco (Lefhaili 2010)
Elevation, slope, aspect	District	----	ASTER Global DEM
Precipitations, temperature (1950 - 2005)	Aggregate values at 5° x 5°	----	Climate Research Unit global dataset
Palmer Drought Severity Index	Aggregate values at 2.5° x 2.5°	---	University Corporation for Atmospheric Research

Table 1: Data

In five villages, household surveys (n=86, sampling rates 30 to 62.5%) provided information on domestic wood consumption. Data on the number of animals per village in 1970 and 2009 were obtained through interviews with local customary authorities (n=6) for 22 villages out of about 70 in the study area. Data on contraventions for illegal wood transport and sales between 1972 and 2007 were provided by the forest officials' registers for 40 villages. The data includes the date of every offense, the name of the forest domain within which it took place, and the origin of the contravener. Direct observation and open interviews provided qualitative insights as to the use and history of forest resources.

3.3 Forest changes

Aerial photographs were geocorrected using the ILWIS program with the ASTER 25m resolution Global Digital Elevation Model (GDEM). The geocorrection was done using the differential rectification method with 20 ground control points (GCP's) per image. The residual Root Mean Square Error (RMSE) on GCP's was 5 to 41 m depending on the image (μ = 15.7 m, stdv. = 11.2 m). Trees were counted at the district level using a set of sample sites (circles 100 m diameter) that were located randomly within the meshes of a grid, with a density of ~1 circle / km^2, excluding built-up areas and oases (1067 circles). A tighter, village-level grid, with a density of ~8 sites/km^2, was used additionally within the territory of the five villages in which the fieldwork was done (n = 622; sampling locations are visible on Figure 1). The sample distribution was identical for the two dates. Topographic variations in some areas caused a slight misalignment of the circle borders, yet the radius of the sample circles was defined to ensure a >90% overlap on average between the areas covered in 1970 and 2007, given the RMSE. The number of trees within each circle, C_{1970} and C_{2007}, was counted visually. Trees on the edge of the circles were counted if their centre fell within the circle. Given the partial overlap of the 1970

2 Data available at http://www.cgd.ucar.edu/cas/catalog/climind/pdsi.html

images, counting was done on the images for which the residual error on the geocorrection control points within a 1500m radius of the sample circle was lowest. The difference *D* between the 1970 and the 2007 values for each circle is the forest density change (Figure 3).

Given the resolution of the SPOT images, it is possible that some trees, especially smaller ones, did not appear clearly on the images and were missed while counting (this was less likely to happen with Quickbird images and aerial photographs, which both have a higher spatial resolution). In order to validate counts on SPOT images, we compared tree counts on the image with on-field tree counts from the 38 sample zones using a linear regression. The regression parameters were then used to correct values counted from SPOT images over the whole area. The corrected value is thus $C_{c,2007}$, and the corrected count difference D_c.

The uncertainty on density change values can be expressed as the sum of several error factors. The *geocorrection error* (*GE*) is due to the residual error after geocorrection. To estimate it, we created a spline interpolation surface on 1970 count values and we computed the slope, which is the maximum gradient value, for each cell. The *GE* for each sample circle was then computed as the product of that slope with the error value of the closest GCP, assuming that this value is similar to the error value at the circle location. The *measurement error* ($ME = ME_{1970} + ME_{2007}$) is due to inaccuracies in the process of counting the trees on images (some trees may be missed while counting, or several trees close to each other may be mistaken to be only one) and was estimated at 10% of counted values. The *correction error* (*CE*) is due to the linear regression used to correct values measured on SPOT images. It was estimated using $CE = a * (C_{c,2007} + ME_{2007})$, where C_{2007} is the 2007 count value, and *a* is a factor equal to half the 90% confidence interval of the regression coefficients. The total measurement uncertainty can thus be expressed as $U = GE + ME + CE$.

To measure the extent to which changes in area estimates reflect degradation as measured by a decrease in tree density, we classified sites as forest or non-forest according to various density thresholds. We then measured the number of sites switching from «forest» to «non-forest» for each threshold value, and compared the results to estimates of forest density change.

3.4 Causes of change

We focused on three main proximate causes of forest cover changes: fuelwood extraction, overgrazing and increasing aridity. The fact that data on different causes were not available at a single spatial scale and for the same observation units did not allow us to conduct a multivariate statistical analysis, although it might have provided interesting insights as to the relative weight of the various causes. Aggregating all the data to the lowest common resolution would have caused a loss of information. We therefore analyzed each factor successively, at the resolution at which each data set was collected.

The relationship between **fuelwood extraction** and forest density decline was tested in two steps. Firstly, we evaluated to what extent domestic consumption might account for the observed decrease in forest density. For this we compared, on one hand, percent change in forest standing volume as predicted on the basis of the rate of domestic wood consumption to natural forest increment to, on the other hand, observed percent forest change $D_{c,\%}$ for the five sub-sample villages. The predicted percent change in forest standing volume for each village was estimated as:

$$D_{p,v,\%} = (100/V_{1970}) * (2007\text{-}1970) * (I_v - W_v)$$

where V_{1970} is standing volume in 1970, I_v is yearly natural increment, and W_v yearly wood consumption (see Appendix A for details on the calculation). A sensitivity analysis was conducted, estimating $D_{p,v,\%}$ with a change of ±50% in the parameters for initial forest standing volume and natural increment. Secondly, we calculated the correlation between D_c in 40 villages and the number of contraventions for illegal wood transport and sales by people from these villages.

The effect of **overgrazing** was estimated based on the correlation coefficient between, on one hand, the mean number of animals per hectare during the period 1970 to 2009 at village level μ_{GD} and, on the other hand, forest change D_c for the 22 surveyed villages.

To investigate the effect of **increasing aridity**, we first analyzed temperature and precipitation data, as well as the Palmer Drought Severity Index (PSDI) for the 1970-2005 period. The PSDI is an index of meteorological drought created by Palmer (1965) based on a supply-and-demand concept of the water balance, and using surface air temperature, precipitation and available soil water content. It has positive values for wet months and negative values for dry months, with values lower than -4 usually considered as a sign of «extreme drought» (details on the computation procedure can be found in Palmer 1965 and in Allen 1984). Monthly values were averaged for years between 1970 and 2005. An increase in aridity would have mostly impacted areas with a low altitude and a high sun exposure, given that these are already subjected to higher aridity on average. We therefore also tested the relationship between forest density change D_c, and sun exposure and elevation on the 1067 sample circles by means of a multivariate, linear regression. Sun exposure was assessed using the Solar Radiation Index (*SRI*):

$$SRI = \cos(\text{latitude})\cos(\text{slope}) + \sin(\text{latitude})\sin(\text{slope})\cos(\text{aspect*})$$

where aspect* is equal to (180° - aspect) (from Keating et al. 2007). We controlled for the effect of initial density by including it as an independent variable in the regression.

4. Results

4.1 Forest change

Forest density in the study area decreased dramatically between 1970 and 2007, from an average of 27.4 trees/ha (stdv. = 13.8) to 15.2 trees/ha (stdv. = 22.8), a change of -12.2 trees/ha, or -44.5% ($p < 2.2e\text{-}16$; see figures 4 and 5). The average uncertainty range on the difference was of 5.7 trees/ha, or 20.1%. The regression of counts on the SPOT image against counts in the field yielded an $R^2_{adj.}$ of 0.84 and a significant estimate (est. = 1.31; $p < 0{,}001$). Figures of percent surface change were much lower than figures of density decrease (Figure 6). Unsurprisingly, the maximum change value (26.3%) occurred when the threshold was set as the average tree density (27.4 trees/ha), as more plots were close to this threshold and were thus more likely to cross it.

4.2 Causes of change

All interviewed households used firewood for baking, and 80% also for cooking. The average fuel mix for the total sample was 4.6 on a 10-point scale, ranging from 0 (only wood) to 10 (only butane). Predicted changes in forest standing volume based on domestic consumption $D_{p,v,\%}$ were positive for three out of five villages, meaning that consumption exceeded regrowth only for the other two villages. $D_{p,v,\%}$ was above $D_{c,\%}$ in all cases except for one (Table 2). Trends were not sensitive to modifications of parameter values for forest standing volume and natural increment by ±50%. Predicted values for village E remained positive for all modifications. Values for villages A and D (1.40% and 3.41%) remained higher than observed values. The value for village B (-29.41%) remained negative and the value for village C remained below observed values in all cases. This shows that domestic wood consumption cannot alone account for the observed decrease in forest density, but that it may have played a role in conjunction with other factors. The correlation between contraventions for illegal wood transport and sales, and forest density change D_c was not significant (c = -0.27; p = 0.096; see Figure 7)[3].

Village	Mean yearly wood consumption per household for 1970-2009 (m^3)	Predicted forest standing volume change (% 1970) $D_{p,v,\%}$	Observed forest density change (% 1970) $D_{c,\%}$
A	14,27	1,4	-27,99
B	10,64	-29,41	-37,42
C	16,22	< -100,0	-26,51
D	14,86	3,41	-49,09
E	11,01	54,5	-15,03

Table 2: Predicted and observed density changes

There was no significant correlation at the village level between browsing intensity, as measured by the average goat and sheep density per village between 1975 and 2009, and forest density change D_c (ρ = 0.22; p = 0.38). Change in sheep and goat density between 1970 and 2009 at the village level was not related to forest density change either (ρ = 0.21; p = 0.38).

There was a significant increase in yearly mean temperature from 1970 to 2005 of about one degree Celsius (c = 0.62; p < 0.001). There was no significant trend for yearly or monthly precipitations. The PDSI decreased significantly from 1970 to 2005 (est. = -0.11; p = 0.018; see Figure 8). A linear regression of *SRI* and elevation against forest density change D_c yielded significant parameter estimates for both variables and an $R^2_{adj.}$ of 0.044 (n=1067). In the most exposed locations, forest density decrease was up to 15 points higher than in the least exposed ones. The effect of elevation was less pronounced, with one point decrease of D_c per 100 m elevation increase. The regression including initial forest density had a much higher $R^2_{adj.}$ (0.6359) and the parameter estimate for initial forest density was positive and significant, which means that denser forests were thinned more than others in absolute terms. While the introduction of this variable changed the magnitude of parameters for *SRI* and elevation (dividing it by two for the first, and by three for the second), it did not change their sign nor affect their statistical significance (see Table 3).

Table 3.: Regression results

3 For correlations, ρ indicates a Spearman coefficient, and c indicates a Pearson correlation.

Variable	Estimate	p-value
Intercept	8,1	0,0039**
Elevation	0,003	0,0035**
SRI	-8,12	0,0009 ***
Initial density	-0,79	<2e-16 ***

5. Discussion

5.1 Forest changes

Our analysis shows that the argan forest in the study area underwent significant density decline between 1970 and 2007. This validates our first claim, that measuring degradation as a change from forest to non-forest systematically underestimated change, with highest deforestation rates just over half as high as density change values. This reflects the fact noted by Rautiainen (2011) that forest cover change detection based on surface measurements is not accurate in cases where tree density is changing. If, as it seems to be the case in many open woodlands (Grainger, 1999), density is declining in dryland forests, forest inventories thus likely underestimate change in these areas.

5.2 Causes of change

Fuelwood extraction for domestic use did not exceed regrowth in 3 out of 5 cases. However, even in these villages, the quantities extracted are hardly insignificant and they may have played a role in forest decline in the long term. Actually, over 40% of the district population living outside forested territory, and people from the nearby market town and from villages on the southern edge of the forest, rely on fuelwood from this area. While gas bottles are increasingly used for cooking, they did not appear in the area before 10 to 20 years ago, and they still provide only part of the energy used. The lack of a significant correlation between contraventions and forest change should be interpreted with caution. It may mean that wood trade has had little or no impact on forest decline, or else that contraventions are not directly related to wood trade intensity. The latter might happen if local forest officials control some areas more strictly than others, or have particular arrangements with locals and traders. What is more, contraventions report the origin of the contravener, not the place where the wood was extracted, which may in some cases be different. Informants acknowledge that most villagers were involved in wood trade until the late 1990's. In dry years, wood sales were the main source of income for many poor households until remittances replaced them as a buffer resource. In the 1980's, wood trade intensified and traders came to the region with pick-up trucks and lorries to supply the cities with wood and charcoal. It would be surprising if this had no impact on the forest. Most locals themselves acknowledge that the cutting has been responsible for part of the forest decline. The relatively low number of contraventions (less than 16 per year on average for the whole study area) suggests that most of the trade was overlooked. No better quantitative index for illegal wood extraction, however, is available.

The absence of any correlation between livestock density and decline in forest density suggests that, contrary to widely-held views, the direct damaging effect of browsing on trees may not be significant compared to other stress factors. Locals also do not seem to believe that browsing has any direct effect on the trees. Our data however does not take nomad herds into account. Saharan nomads and, more recently, nomads from Eastern provinces have used the less mountainous parts of the study area intensively. Their herds are often greater than those of the local communities. Unfortunately there are no statistics on nomad movements and their herd size in the area. Likewise, the effect of grazing on regrowth and on soil and understory degradation could not be evaluated directly. We saw no saplings in the area during fieldwork. When asked about it all informants, including older people, replied that they had never seen argan saplings growing to become mature trees. Some acknowledged that the few saplings that did grow were eaten away by goats, but others said that regrowth solely depends on rains.

Temperature increased during the study period, and while there appeared to be no trend in rainfall, the Palmer Drought Severity Index decreased significantly, suggesting a regional increase in aridity. Low and exposed surfaces were impacted most, as trees disappeared at rates much higher than on other surfaces. This supports the hypothesis that aridity impacts trees, and that part of the observed decline is to be attributed to increasing aridity. It is not clear whether local human activities such as grazing have had any influence on the vulnerability of trees to drought, as posited in the conceptual model. It is not clear either whether increasing aridity is to be attributed to human-induced climate change or to long-term climatic oscillations such as the ENSO (as argued by Esper et al. 2007). Prospects of an increasingly arid climate in North Africa in the 21st century (IPCC 2007) give reasons for concern.

It is clear from this analysis that social and biophysical, and local and distal drivers all play a role in forest density decline and cannot be viewed in isolation in the argan woodlands. This validates our second claim, on the need for an integrated approach to the understanding of causes of dryland forest degradation. The difficulty inherent to this type of systems is that of measuring the relative weight of these drivers. As we have shown, data on drivers of change are difficult to acquire and analyze at a single scale. Moreover, these causes act in a synergistic rather than in an independent manner. We suggest that more research is needed to design methodologies that can make good use of available data to analyze interactive causal clusters of changes in dryland forests while explicitly addressing the multi-scale nature of these processes.

6. Conclusion

We argued that dryland forests are undergoing high rates of change that make it necessary to build a better understanding of the dynamics of land-cover change in these ecosystems. This was confirmed in the case of the argan woodlands in Southwest Morocco, which underwent a dramatic density decline of over 40% between 1970 and 2007 in our study area. We demonstrated that monitoring forest change based on area estimates, as done for tropical forests, is inappropriate for dryland forests, because density change may be more important than surface change. Indeed, in the case of the argan forest, surface data strongly underestimate change, independently of the density threshold used to distinguish forest from non-forest. As for the causes of change, we argued that causal explanations derived from the understanding of tropical deforestation or desertification, and emphasizing either local or remote, and either social or biophysical drivers, cannot satisfactorily account for processes involved in dryland forest degradation. The existing literature on dryland forests still tends to be polarized along four discourses that emphasize neo-malthusian, climatic, economic, or political ecology explanations. Our case study showed that degradation processes transcend these categories: the decline in forest density can only be explained by a combination of factors, including increasingly dry climatic conditions, fuelwood demand from both local villages and remote markets, and possibly, grazing pressures by the herds of nomads, even though this latter factor could not be evaluated based on our data. Both social and biophysical drivers are at play, and most of these drivers have remote as well as local components. Livelihood choices for instance, such as selling wood or increasing the size of goat herds, are influenced by the availability of alternative livelihood sources. Our subsequent work on livelihoods in the study area shows that migration, remittances and non-farm opportunities have shaped the development of local communities. These factors are in turn determined by higher-level political choices and economic opportunities.

The fact that aridity seems to be a major driver of forest decline in the argan woodlands has important policy implications. Firstly, although it may be necessary to impose tighter restrictions on forest use, including firewood harvesting, this may not be sufficient to prevent forest decline. Indeed, the IPCC has predicted an increase in aridity over the 21st century in North Africa (IPCC 2007), which would imply a corresponding decline in forest density. If there is a feedback loop between loss of density and worsening local-level aridity, as suggested in Figure 1, increasing density over a certain threshold by planting and protecting new trees might reverse the trend, making micro-climatic conditions suitable again for germination and sapling survival. This remains to be tested however. Secondly, the role of aridity raises questions as to the sustainability of forest-based livelihoods. There has been much hope in the last decade that the high prices for argan oil would allow poor households in the region to improve their living conditions and that argan oil production, particularly through women cooperatives, would become the new sustainable livelihood. However, argan harvests depend on rains and on the quality and density of the forest. Many surveyed households reported that they had not benefited from the boom because forest density had decreased and their harvest had been negligible in times of high prices - as with any market commodity, prices are highest when the resource is scarce. Prospects for continuing forest decline and increasing aridity should temper excessive optimism on the contribution of argan oil to local rural development on the long term.

As for any case study, our findings have limited validity outside their geographical scope. However, this study suggests three conclusions of general interest. Firstly, as shown in other case studies reviewed in the introduction, dryland forests may be receding in places at rates comparable to some tropical forests. A consistent picture emerges from the accumulation of local case studies that call for greater attention to processes of degradation in arid and semi-arid forests and woodlands. Secondly, although standard monitoring methods tend to be used to estimate change in dryland forests, the methods currently used are probably inappropriate and need to integrate changes in forest density. In this study, we implemented a simple and inexpensive method for monitoring forest density that could easily be applied to other regions. Thirdly, explanatory frameworks derived from the literature on desertification and tropical deforestation are not perfectly appropriate for dryland forests. A more integrated understanding of the causes of dryland forest degradation needs to be developed, taking into account the full complexity of these multifunctional landscapes, including biophysical and social, and local and distal components.

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Appendix A : Details on the calculation of domestic wood consumption

Total wood consumption per village between 1970 and 2009 was estimated by extrapolating household means to the whole population, taking into account population changes and changes in the fuel mix between the two dates (wood consumption before the adoption of butane was increased proportionally to the quantity of butane used today). Initial wood volumes in the forest were estimated using the 1990 mean standing volume for the argan forest (18m3/ha) as published by forest officials (Lefhaili, 2010). It was assumed that the 18m3/ha value reflected the 1990 mean for the five villages. This value was augmented to reflect the average change in forest density from 1970 to 1990, assuming a linear change from 1970 to 2007. The resulting value was then multiplied for each village by the deviation from the mean forest density, so as to reflect local differences. Natural increment was estimated at 0.3m3/ha (Lefhaili, 2010) and adapted in the same way. As for the forest change values at village level, village territories were given the mean of the forest density values for the sample circles falling into them. Village boundaries were delineated using Voronoi diagrams (Voronoi diagrams have edges that are at equal distance between each point and the next closest point, which is similar to reality, because village borders are normally at approximately equal distance from neighboring villages). There were on average 13.3 sample circles in each village territory (stdv. = 6.3).

Figures

Figure 1 : Conceptual framework

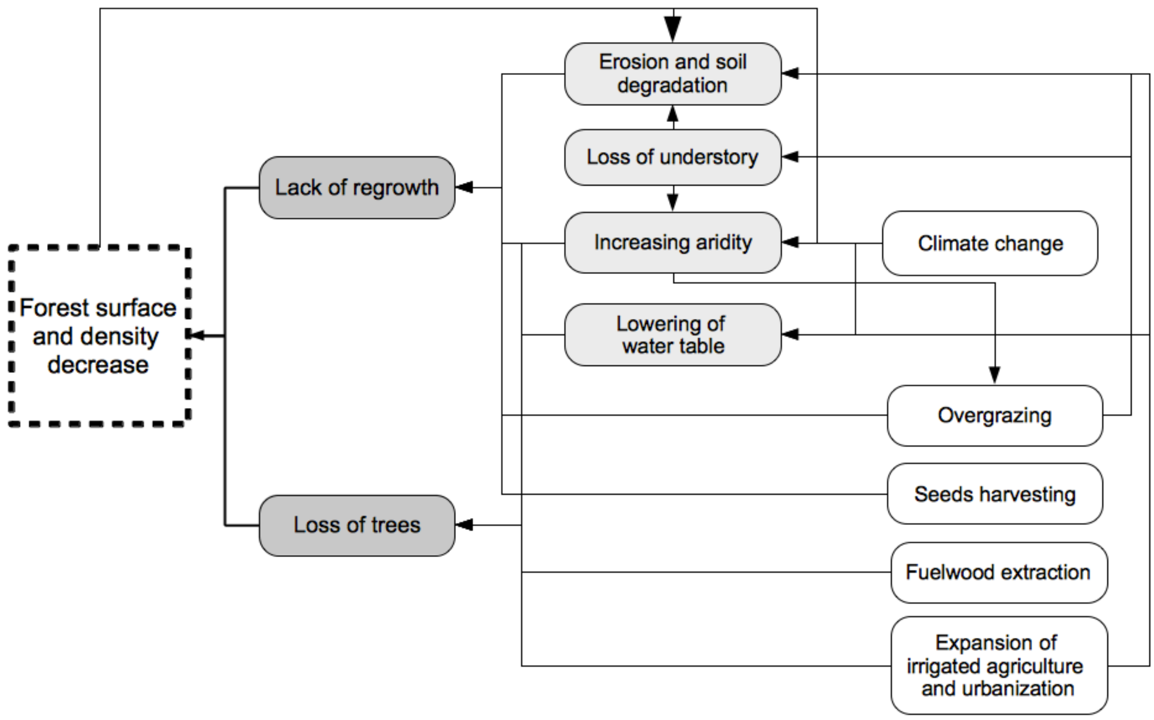


Figure 2 : Study area and sampling

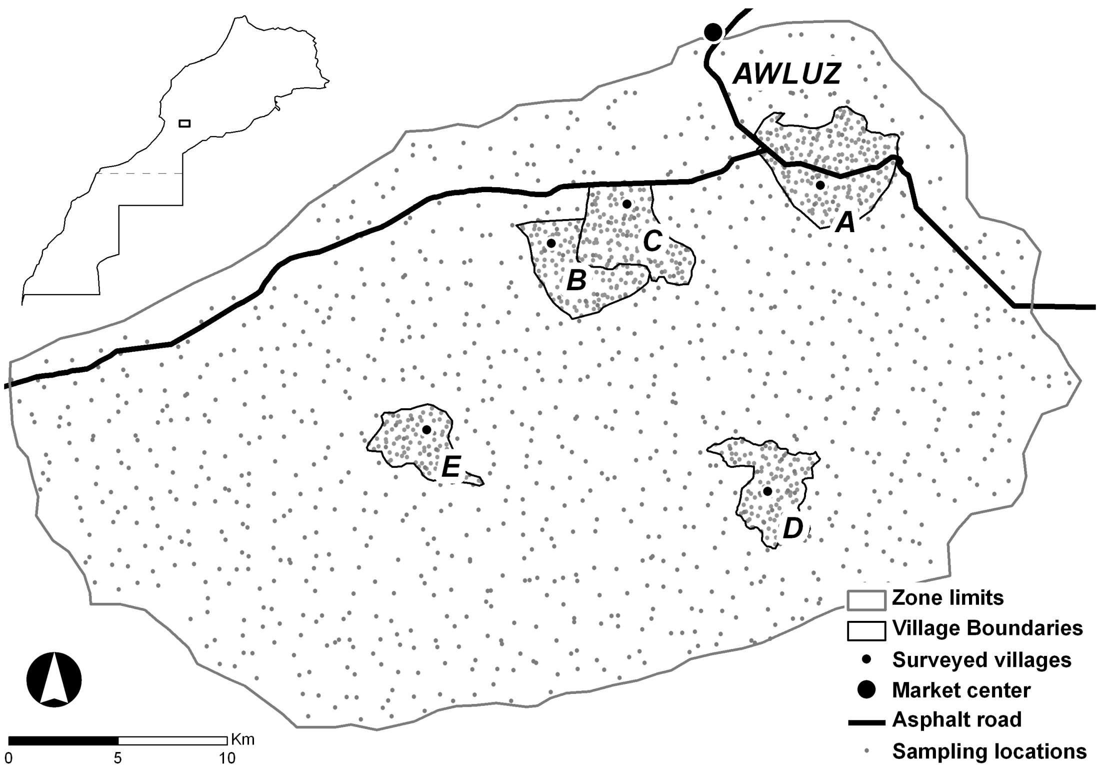


Figure 3 : Counting method

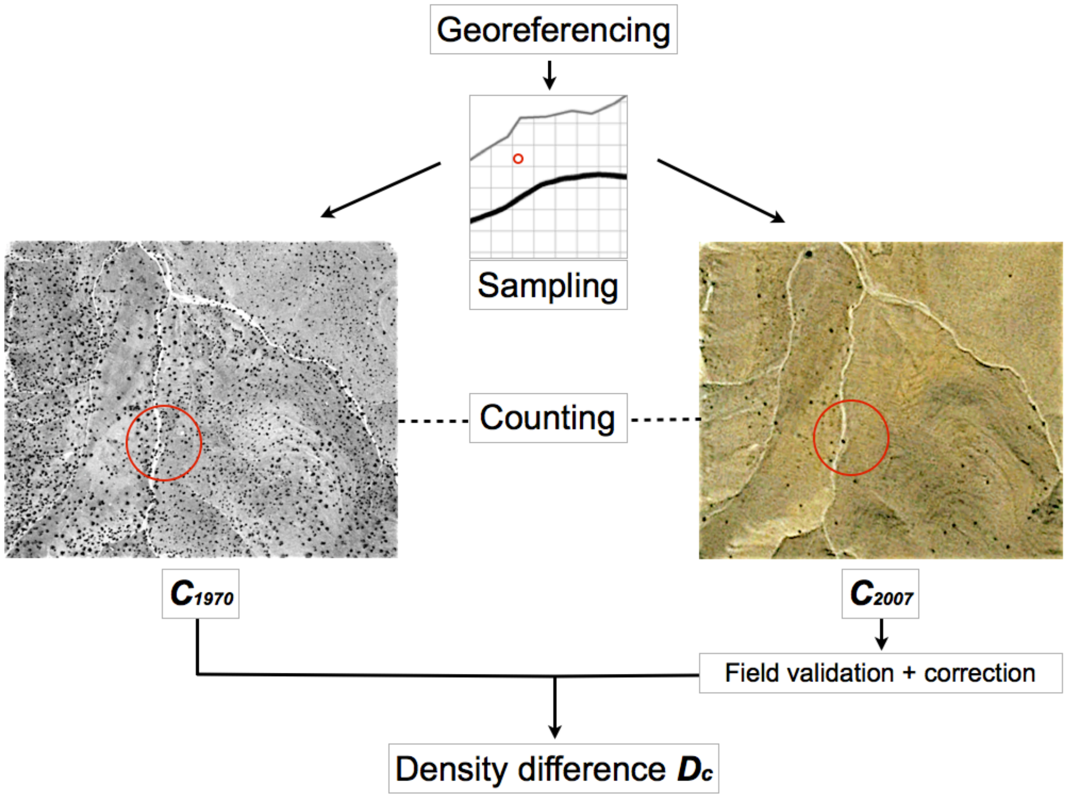


Figure 4 : Woodlands density in 1970

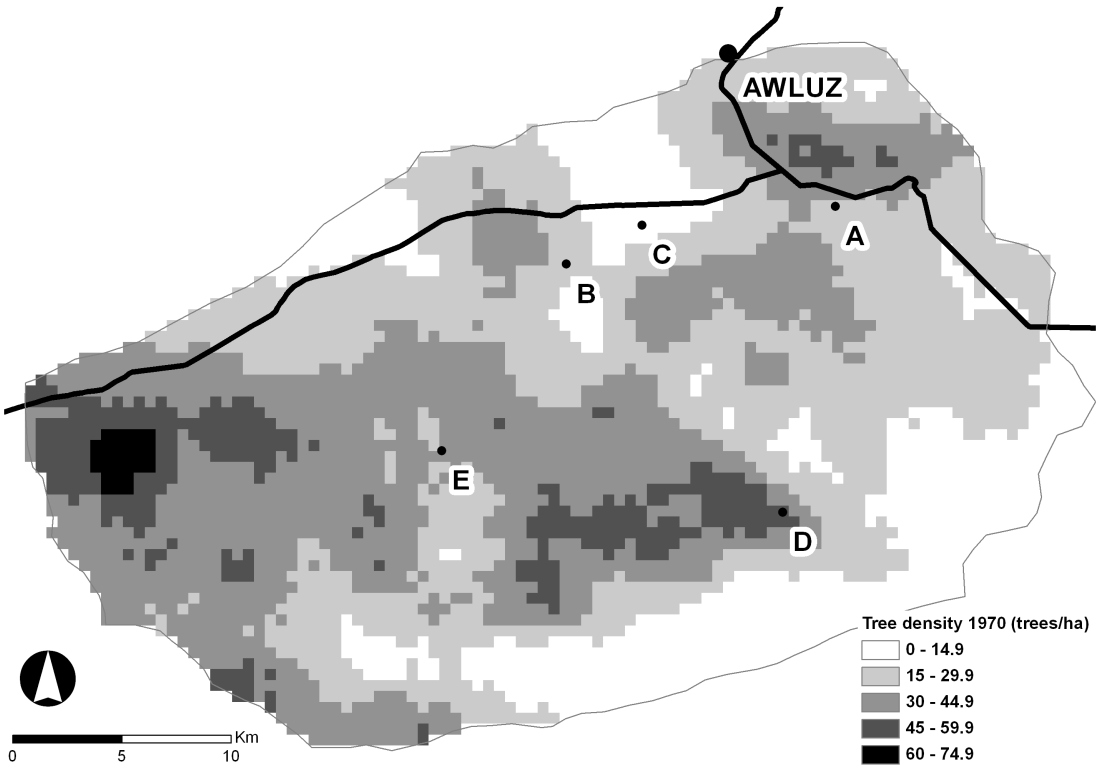
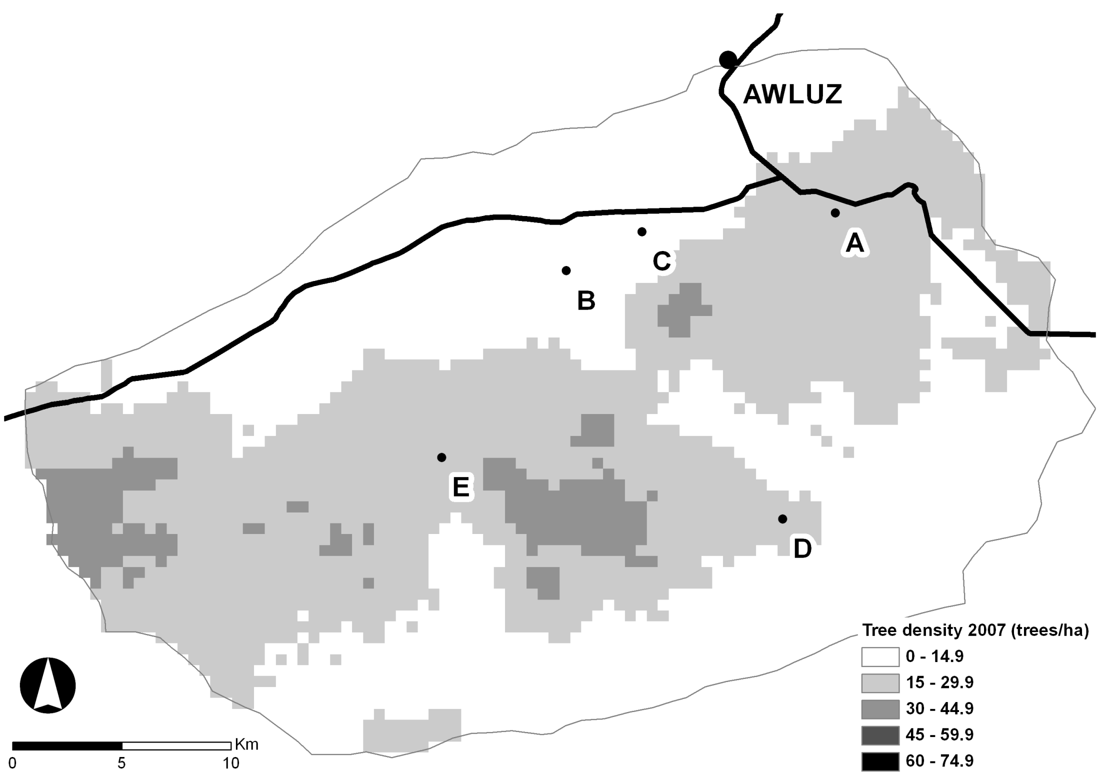


Figure 5 : Woodlands density in 2007



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