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Role of Deforestation on Spatial Variability of Soil Nutrients in a Hyrcanian Forest

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ABSTRACT Studying forest degradation through evaluation of soil nutrient concentrations, which reveals soil functioning within the ecosystem, is necessary for sustainable management of land resources. This research was conducted to understand the changes of soil nutrient, resulting from exploitive management using some soil features and their spatial pattern. In the mid-summer of 2014, two sites were selected consisting of an undisturbed forest site (FS) and a completely deforested site (DS); both sites were in lowland part of Khanikan forests located in Mazandaran Province, north of Iran. Within each site 50 soil samples were obtained from 0-30cm depth along two sampling lines with 500 meter length thus resulting in 100 soil samples for each site. The interval between samples along lines and also the distance between lines were selected 10 m. The mean pH was lower at the DS (5.70) than FS (6.58). The mean of soil organic carbon (SOC) was significantly higher at FS (2.78 %) when compared with DS (0.56 %). Total nitrogen (N) also followed the same trend having significantly higher values at FS (0.28%) than DS (0.16%). Mean available phosphorus (P) values were significantly higher at the FS (17.33 mg kg^{-1}) than at the DS (7.24 mg kg^{-1}). The amounts of available potassium (K) were significantly higher at the FS (148.15 mg kg^{-1}) than DS (84.14 mg kg^{-1}). A geostatistical analysis revealed that deforestation changed the spatial variability models and fractal dimension of soil features. As a conclusion, the spatial variability of soil pH and SOC were more imposed by deforestation compared to the other soil features. Our results suggest that deforestation should be regarded as an effective factor on variability of soil nutrient that are tied to forest ecosystem management.

Key words: *Forest degradation, Fractal dimension, Geostatistics, Soil chemistry*

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1 INTRODUCTION

Forests around the world have undergone severe disturbances due to anthropogenic factors. An ever increasing human population has migrated into forested zones and cleared the forest to facilitate economic activities such as farming, grazing, and establishment of settlements and industries (Dinesh *et al.*, 2003). Destroying forest and rangelands and changing them into agricultural and residential lands, land-use change, have been very noticeable particularly in the northern part of Iran because of agricultural activities and development of human societies. Land-use changes in Iran have

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been more rapid in the past 50 years than at any time in Iran's history and are expected to continue or accelerate in the future. With a rapidly increasing population and a strong rise of the standard of living, the need to intensify agricultural production increased; this situation puts pressure on other resources. The natural forests in Iran have been reduced from 19 million hectares in the 1950s to 12.4 million hectares in the 1990s. During the past 50 years, the amount of Iran's cultivated land has grown by more than five times, increasing from 2.6 million hectares to 24.5 million hectares (Bahrami *et al.*, 2010).

The growth process of urban societies has been increased all over the world and it is predicated that it will have been increased up to 60% by the year 2030 (McGee, 2001). Deforestation can be generally defined as: the reduction of the capacity of a forest to provide goods and services (Miettinen *et al.*, 2014). However, this general definition can be interpreted in numerous and potentially rather contradicting ways. Depending on the scope of the analysis, the evaluation of degradation in a given forest site can be based on (1) biological diversity; (2) forest health and vitality; (3) productive functions of forest resources; (4) protective functions of forest resources; and (5) socio-economic functions of forests (Miettinen *et al.*, 2014). Deforestation has many significant ecological consequences. The removal of vegetation results in increased erosion of soil sediments, which are many times deposited in water bodies, consequently depositing soil particles and nutrients. A decrease in vegetation also corresponds with a decrease in nutrient uptake in the soil, resulting in an increased rate of nutrient leaching from the soil (Ketterings *et al.*, 2002; Page *et al.*, 2005; Gassman *et al.*, 2006; Maloney *et al.*, 2007; Huang *et al.*, 2007). The leached nutrients are often deposited in water bodies. Both types of nutrient inputs subsequently alter physical stream characteristics as well as rates of productivity and ecological components of water bodies affected by deforestation (Burnsa *et al.*, 2005).

Deforestation is the most dynamic driving factor of terrestrial carbon stock changes, soil organic carbon (SOC) storage, and also an important factor in future carbon sequestration that cannot be ignored (Ussiri *et al.*, 2006; Poeplau and Don, 2013). The variability of SOC amounts imposed by deforestation could become crucial in terms of future policies to mitigate the global greenhouse effect (IPCC, 2000; Wang and Fang, 2009). Therefore, it is critical to understand how SOC varies in response to forest degradation when evaluating the role of terrestrial ecosystem processes in altering the global carbon cycle and carbon accumulation in the atmosphere (Jiao *et al.*, 2010; Mishra *et al.*, 2010). Compared to other soil nutrients, soil N, P and K are considered essential nutrients that most frequently limit soil productivity (Giesler *et al.*, 2002; Huang *et al.*, 2007) and soil microbial activity (Liu *et al.*, 2010). Furthermore, soil N, P and K levels are closely correlated with SOC cycles (Bronson *et al.*, 2004), which have dynamic effects on greenhouse gas emissions that are linked to global climate change (Lal, 2004). Thus, a better knowledge of soil N, P and K levels and their distributions is necessary when evaluating current or potential soil productivity and assessing potential environmental pollution, as well as for a better understanding of climate change and its feedbacks (Jennings *et al.*, 2009).

Spatial heterogeneity is a common characteristic of soils and the spatial variability of the physical and chemical properties has been a topic of major concern to soil scientists (Wang *et al.*, 2010; Chuai *et al.*, 2011; Tesfahunegn, 2014). Understanding the spatial variability of soil features is the key to the understanding of the landscape-scale processes of soils (Corwin *et al.*, 2006). There have been

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growing interests in the study of spatial variation of soil features using geostatistics since 1970s, as geostatistics were well developed and successful in characterizing the spatial variations of heavy metals (Yu *et al.*, 2001; Romic and Romic, 2003), soil nutrients (Liu *et al.,* 2004; Aishah *et al.*, 2010; Jing *et al*., 2014; Saglam and Dengiz, 2014) and other soil characteristics (Lima *et al.*, 2008; Gilbert and Wayne, 2008; Liu *et al.,* 2008; Siqueira *et al.*, 2014). The findings of Chen *et al.* (2006) indicated that the spatial distribution pattern of the SOM, total P and K had a longer range of spatial dependency than in soil total N. Like other soil properties, nutrient elements are distributed heterogeneously in soils, and the degree of variation is a function of the study scale and/or its aspects (*e.g.* support, spacing, and extent) (Wang *et al.*, 2009). This spatial heterogeneity is caused by various factors (Jenny, 1941), which include climatic variables (Patil *et al.,* 2010), parent material (Lin *et al.*, 2009), topography (Rezaei and Gilkes, 2005), vegetation types (Rodríguez *et al.,* 2009), soil texture (Gami *et al.,* 2009) and deforestation (Meersmans *et al.,* 2008; Rodríguez *et al.*, 2009; Wang *et al.*, 2010; da Rocha Junior *et al.*, 2014).

Thus, from the perspectives of ecosystem research and environmental protection, it is important to know the spatial variability of soil nutrient as affected by deforestation in the degraded site (Schöning *et al.*, 2006; Wang *et al.*, 2009; Rodríguez *et al.*, 2009) of northern Iran. Therefore, the objectives of our study were: (1) to investigate the current status and spatial variability of soil nutrients in surface soils across the degraded and adjacent intact forest in a lowland part; (2) to provide an overview of the regional distributions of soil nutrients and to calculate the fractal dimension in a deforested site and adjacent intact forest using a geostatistical method.

2 MATERIALS AND METHODS

2.1 Study site

This research was conducted in Khanikan forests, with an site of 2807 ha, that is located in the lowland and midland of Mazandaran Province in north of Iran, between 36˚ 33́ 15″ N and 36˚ 37′ 45″ N latitudes and 51˚ 23′ 45″ E and 51˚ 27′ 45″ E longitudes (Figure 1a). The elevation of the forest site ranges between 50 and 1400 m above sea level (a.s.l.). Minimum temperature in December (7.5˚C) and the highest temperature in June (24.6˚C) are recorded, respectively. Mean annual precipitation of the study site were from 47.5 mm to 237.6 mm at the Noushahr city metrological station, which is 10 km far from the study site. The climate is temperate moist, on based of Demarton classification, and the dry months extend from May to September. The soil is forest brown soil showing a texture that ranges between sandy clay loam to clay loam as showed in figure 2. (Mollaei-Darabi *et al.,* 2014). The soil order name is Alfisols. The study site is on uniform terrain with 200-230 m a.s.l., moderate slope (30-35%) and north exposure. The dominant forest types included Hornbeam (*Carpinus betulus* L.) and Persian ironwood (*Parrotia persica* C. A. Meyer) (Mollaei-Darabi *et al.,* 2014). A lowland part of these forests, almost 7 ha, were destroyed because of extensive exploitation carried out by local residents about 30 years ago (Figure 1b).

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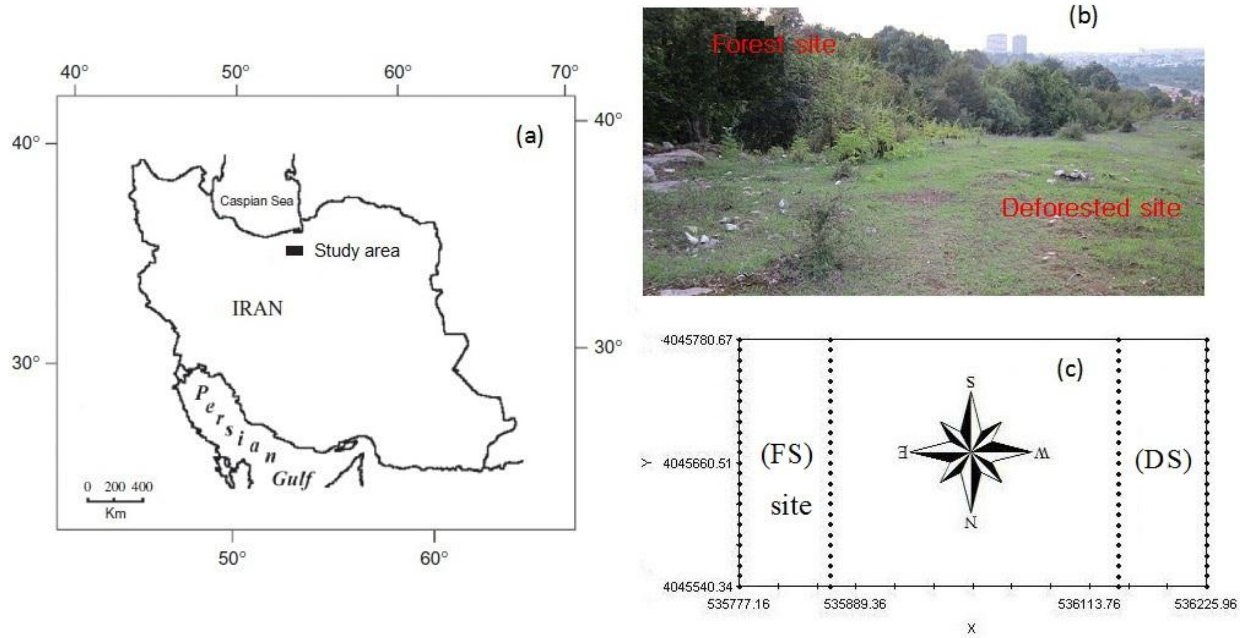

Figure 1 Study site located in Mazandaran Province (a and b), north of Iran and Schematic representation of the experimental design (c) (figure not to scale) adopted for soil sampling pattern in forest (FS) and deforested sites (DS)

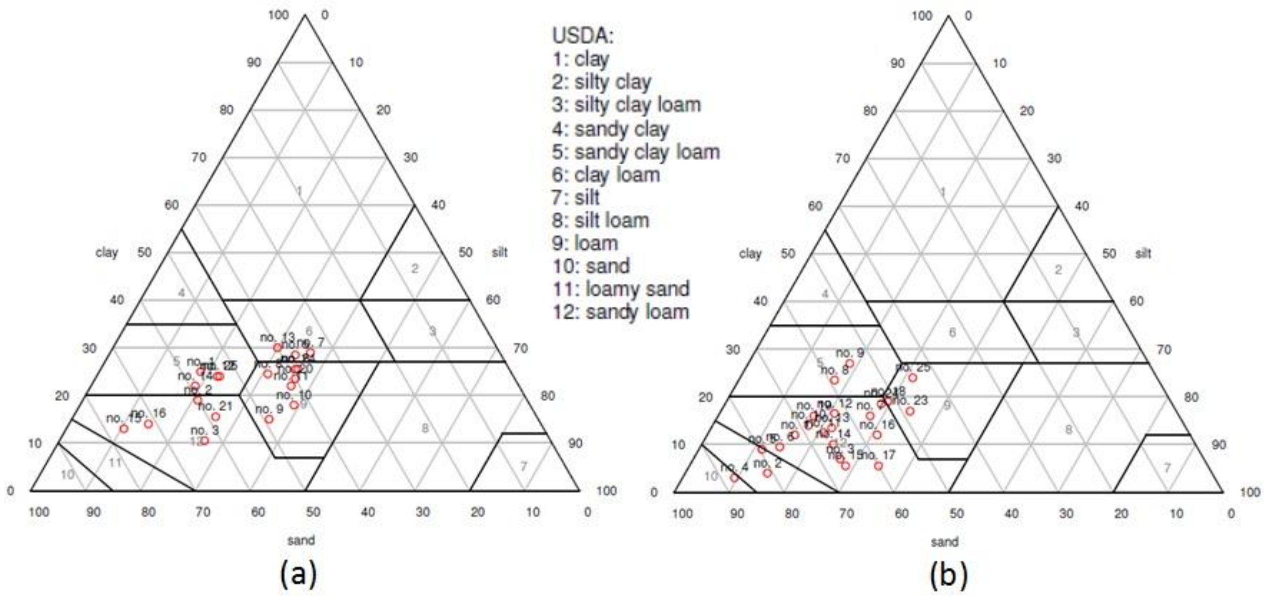

Figure 2 Soil texture classes in forest (a) and deforested (b) sites

2.2 Soil sampling and analysis

Two sites with about 300 m apart from each other were selected consisting of an undisturbed forest site (FS) and a completely deforested site (DS). Within each site 50 soil samples were obtained from 0-30 cm depth along two sampling lines with 500 meter length thus resulting in 100 soil samples for each site. The interval between samples along lines and also the distance between lines were selected 10 m (Figure 1c). Soils were air-dried and passed through 2-mm sieve (aggregates were broken to pass through a 2 mm sieve). Active soil reaction (pH) was determined using an Orion Ionalyzer Model 901 pH meter in a 1:2.5, soil: water solution (Kooch *et al.*, 2015). Organic C was determined using the Walkey-Black technique (Allison, 1975). Total N was determined using the Kjeldhal method (Bremmer, 1960). Available P was determined with spectrophotometer by using Olsen method (Homer and Pratt, 1961). Available K by ammonium acetate extraction at pH 9 was determined with Atomic absorption Spectrophotometer (Bower *et al.,* 1952).

2.3 Statistical and spatial analysis

Descriptive statistics for each soil variable was computed by the software SPSS for Windows Release 17.0. Prior to the geostatistical data analysis, Kolomogorov-Smirnov test was used for testing normality and Levene test for data homogeneity testing. Independent sample t-test was used to find differences in soil features between the two sites. An experimental semi-variogram was developed to determine the spatial dependence of soil features using the following equation (Trangmar *et al.*, 1985):

$$\gamma(h) = \frac{1}{2N(h)} \sum_{i=1}^{N(h)} \left[z(x_i) - z(x+h) \right]^2 \quad (1)$$

Where γ *(h)* is the semi-variance; *N (h)* the number of experimental pairs separated by a distance; and $z(x_i)$ the measured sample value at point x_i.

The autocorrelation structure is depicted by the variogram, the central tool of geostatistics. The spatial structure of each variable has been defined from the semi-variogram parameters, *i.e.* nugget, sill (or total semi-variance) and range. The nugget is the variance at the zero distance and represents the experimental error; sill is the semi-variance value at which the semi-variogram reaches the upper boundary after its initial increase (Cressie, 1993). This variance is the maximum for this kind of semi-variogram and represents the total semi-variance of the study site; range is the value (x-axis) at which one variable becomes spatially independent, being the lag-distance at which the semi-variogram flattens. The nugget to sill ratio quantifies the importance of the random component and provides a quantitative estimation of the spatial dependence (Isaaks and Srivastava, 1989). The theoretical model coefficients of the semi-variogram were determined by the fitness of the mathematical model to the γ *(h)* values. The following models were fitted to the data:

(a) spherical, $\gamma(h) = C_0 + C_1 [1.5 (h/a) – 0.5 (h/a)^3]$, for $0 > h < a$, and $\gamma(h) = C_0 + C_1$ for $h > a$; (2)

(b) exponential, $\gamma(h) = C_0 + C_1 [1 – \exp(–3h/a)]$, for $0 < h < d$, (3)

where C_0 is the nugget effect, $C_0 + C_1$ is the sill, a is the range and d is the maximal distance in which the semi-variogram is defined. To determine the spatial dependence, the semi variogram was examined by using the program Gs+ version 9 (Gamma Design Software, LLC, Plain well, MI).

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Spatial class ratios (relative nugget) similar to those presented by Cambardella *et al.* (1994) were adopted to define distinctive classes of spatial dependence. A variable is considered to have a strong spatial dependency if the ratio is less than 25%, moderate spatial dependency if the ratio is between 25-75% and weak spatial dependency if the nugget/sill ratio is greater than 75% (or pure nugget; *i.e.* when slopes of semi-variograms are close to zero). Kriging is a procedure for estimating regionalized variables at unsampled points, based on initial data value. However, ordinary kriging, the workhorse of geostatistics, is the most common type of kriging in practice, particularly in environmental sciences (Webster and Oliver, 2000). It is given by:

$$Z(x_0) = \sum_{i=1}^{n} \lambda_i Z(x_i) \qquad (4)$$

where, λ_i is the weight associated with each sample location value. To evaluate the results of kriging usually, a Jack-knife cross-validation approach is used. All the samples are excluded one by one from the data set and estimated again by kriging using the remaining samples. Then measured data and estimated values are compared to evaluate the kriging results (Webster and Oliver, 2000). In this study, the accuracy of kriging is measured using Mean Bias Error (MBE) and Mean Absolute Error (MAE) as below:

$$MBE = \frac{\sum_{i=1}^{n} (R_S - R_O)}{n} \qquad (5)$$

$$MAE = \frac{\sum_{i=1}^{n} |R_S - R_O|}{n} \qquad (6)$$

where, *rs* is the estimated sample value at point; *Ro,* is the measured sample value at point xi; *n* is number of samples. The most accurate prediction was indicated by the smallest MBE and the MAE close to zero (Webster and Oliver, 2000). Furthemore, the surface variogram was employed for investigation of isotrophic condition and Jack-Knife cross-validation approach was used to evaluate the results of kriging (Webster and Oliver, 2000).

3 RESULTS AND DISSCUSSION

The soil variability data in Table 1 show that all the CVs were between 1.16% and 45.81%. These outputs indicate that soil chemical features in the study site were little to moderately variable at the local scale according to Nielsen and Bouma (1985).

Table 1 Descriptive statistics for soil features in study site

Variables	pH		Organic carbon (%)		Total N (%)		Available P (mg kg^{-1})		Available K (mg kg^{-1})	
	FS	DS	FS	DS	FS	DS	FS	DS	FS	DS
N	100	100	100	100	100	100	100	100	100	100
Mean	6.61a	5.93b	2.78a	0.56b	0.28a	0.16b	17.33a	7.24b	148.15a	84.14b
Std deviation	0.37	0.48	0.46	0.25	0.06	0.07	2.53	1.51	32.64	20.64
Min.	5.88	4.96	1.09	0.17	0.01	0.05	11.56	4.50	59.09	37.49
Max.	7.12	6.90	3.78	1.09	0.39	0.32	24.34	11.45	178.98	112.78
Skewness	-0.32	0.14	-0.98	0.57	-0.97	0.85	0.26	0.52	-0.84	-0.60
Kurtoisis	-1.17	-0.80	1.46	-0.95	1.77	-0.39	0.41	-0.32	-0.67	-0.80
CV (%)	5.62	8.20	16.58	45.81	24.11	44.09	14.60	1.16	22.03	24.53

* Contrasting letters a, b refer to significant differences between forest site (FS) and deforested site (DS)

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The soil reaction was slightly acidic at both the sites. The mean pH was lower at the DS (5.70) than FS (6.58) (Table 1). Higher acidity under deforested systems might have favoured solubilization and removal of cations in leaching water. Low temperatures can limit microbial decomposition of SOM (Karhu *et al.*, 2010), leading to accumulation of nutrient elements in soils of FS (Leifeld *et al.,* 2005) than in DS. The differences between soil fertility in FS and DS can be primarily explained by a severe decrease of SOM input to the litter once the forest is extracted from the system. It is also important to highlight the intensity of rainfall in this northern Iran region. Precipitation in this region undeniably plays a crucial role in fertility loss through nutrient leaching. This phenomenon is well illustrated by the nutrient elements decrease of up to more in DS (Zheng, 2005). Soil texture is of extreme importance as it has a great influence on the chemical features of soil, so should be considered when interpreting results (Ashman and Puri, 2002; Brady and Weil, 2008). Soils dominated by clay particles, and those which contain high SOM content, FS in our study site according to Mollaei-Darabi *et al.* (2014), therefore have a higher cation exchange capacity compared to sandier soils (those collected from the FS), which comprise of little negatively charged colloidal material (Brady and Weil, 2008; Holden, 2008). This could explain the significantly higher concentration of exchangeable base cations observed at the FS, compared to the DS.

As previously established, soil texture plays a huge role in the loss of exchangeable base cations from a soil. Soil collected from the DS, consisting of a sandy clay loam (according to Mollaei-Darabi *et al.,* 2014 findings), would pose larger pore spaces than that of the FS, thus allowing rapid water movement through the soil, and permitting dissolved exchangeable base cations to be easily lost (Gerrard, 2000; Ashman and Puri, 2002). The decrease in exchangeable cations could also be explained by the increase in soil acidity. Exchangeable base cation concentrations would decrease, as exchangeable aluminum (Al) and exchangeable hydrogen ions would likely dominate cation exchange sites (Brady and Weil, 2008; Berthrong *et al.,* 2009). This was also found to be true by Johnson and Lindberg (1989), who concluded that a decrease in exchangeable base cations and increase in soil acidity were caused via acid deposition (Richter and Markewitz, 2001; Olszewska and Smal, 2008).

Soil pH is a master variable indicative of many aspects of soil chemistry, and controls the availability of nutrients for plant uptake (Ludwig *et al.,* 2001). Soil reaction was significantly higher in FS compared to the DS. This does, however, very much indicate that the removal of trees has been an influential factor in reducing soil acidity. Neal *et al.* (1992) also observed that tree canopy removal in Wales led to a decrease in soil pH, which they suggested was due to a dramatic reduction in the capture of acidic pollutants. Accordingly, Reiners *et al.* (1994) reported the decrease in acidity after conversion of tropical forest to pasture and subsequent abandonment in the Atlantic Zone of Costa Rica. Aluminum (Al) saturation on the soil exchange complex declines following forest degradation (Numata, 1999). Higher Al under degraded site could also be attributed to higher acidity and this was supported by the strong negative correlation between Al and pH (Adam *et al.,* 2001; Chen *et al.,* 2001).

The mean of SOC was significantly higher at FS (2.78 %) when compared with DS (0.56 %) (Table 1). The SOC acts as an exchange surface for cations and direct source of N, P, and (S) through microbial carbon (C) and N-immobilization/mineralization reactions (Rasiah and Kay, 1999). The level of SOC is positively correlated with the total SOM and various formulas exist from which one can use SOC to

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estimate total SOM (Buol *et al.,* 2003). SOC and SOM are added to the soil primarily from decomposing vegetative residues such as leaves, litter, and roots and a decrease in these inputs can lead to a decrease in SOC and SOM (Bernoux *et al.,* 1998). SOM accumulated in forest floor can be reduced by erosion (Abbasi and Rasool, 2005; Zheng *et al.,* 2005). The forest top soil (0-30 cm) had the highest amount of SOC (2.78 %) compare to the DS at the same depth (0.56 %). A comparison of soil profiles in DS and FS showed that 30 cm of top soil have eroded away in the DS. The SOM that currently forms the A horizon in the DS originated from the decomposition of grass litter and roots. In contrast, the SOM in the upper layer of the forest soils is the result of decomposition of roots and litter fall from trees, shrubs and herbs. The total SOC content in the forest ecosystems is greater than in the DS due to higher carbon concentrations and thicker A and B horizons. Significant decrease in SOC storage of our study site as a result of deforestation agrees with Yimer *et al.* (2007); Nourbakhsh (2007); Smal and Olszewska (2008) and Khresat *et al.* (2008) results.

Total N also followed the same trend having significantly higher values at FS (0.28%) than DS (0.16%) (Table 1). Considering that soil N directly originates from the plant litter, it might be expected that soil N contents would not show significant difference. However, soil N contents of FS were significantly different from the DS. Forest degradation brought significant changes to total N. The loss of N from deforestation sites appear to be particularly important. DS at surface (0-30 cm) had at least half as much as total N than forest soil. The largest amount of total N (same as SOC) was observed for the forest top soil (0-30 cm). Patrick and Smith (1975) reported that total tree harvesting caused the nutrient, including N, to be removed up to three times compared to conventional lodging. In addition to losses from biomass removal, nutrient can be lost from deforested sites by increased soil nutrient mobilization and leaching, when little vegetation is present to take up (Mroz *et al.*, 1985). Rasmussen (1998) reported that all deforestation may induce some N leaching during a short period before revegetation by herbs or trees.

The higher levels of total N in the FS compared to the DS can be explained by the higher amounts of SOM in the forest soils. In addition, removing the overstory in the forest site caused probably increase of SOM decomposition and N transformation rates resulting in more N being leached out of the soil (Khresat *et al.,* 2008). Similar to the forest N levels in this study, Mroz *et al.* (1985) found total N to decrease following whole tree clear cuts on three sites. In another study where erosion was an important driver of site characteristics. Zheng *et al.* (2005) reported that erosion following deforestation resulted in a 46.7% decrease in total N. Similar to our results, Abbasi *et al.* (2007) recorded total N levels in degraded site to be nearly half those in forest lands. Based on a study in Ethiopia, Yimer *et al.* (2007) also found deforested regions to have lower total N levels than native forests. Many other studies have found the opposite result. Glaser *et al.* (2000) and Savozzi *et al.* (2001) reported grasslands and pastures to have higher levels of N than forestland. The disparity I these findings with regards to the results found in this study are probably due in part to the low levels of SOM in the pasture and point to its overgrazed and degraded nature.

Mean available P values were significantly higher at the FS (17.33 mg kg^{-1}) than at the DS (7.24 mg kg^{-1}) (Table 1). Phosphorus is primarily introduced to a soil via organic matter (Abbasi *et al.*, 2007; Brady and Weil, 2008). As would be expected, the FS, which had higher levels of SOM than the DS, also had higher levels of available P than the DS. Available P in soil decreased with the deforested that it could be attributed to the erosion rate caused by per unit of rainfall erosivity that increased with the deforested. Weathering of primary minerals is the main source of P to ecosystems, but in highly acidic systems, P can be quickly immobilized and made inaccessible by complexation with organic or Fe-Al compounds, creating local complex patterns. Different studies have recorded various changes in P levels due to deforestation and land cover change. Similar to the relative values of the forest, Zheng *et al.* (2005) found deforestation and subsequent erosion to result in an 86.6% loss of P. In contrast, Boyle *et al.* (1973) reported no long term loss of P due to whole tree clear cutting in Wisconsin but did note that overall nutrient losses may have been greater if the cut was on a steeper slope or if the regeneration following the cut was suppressed. Nearby, in the Upper Peninsula of Michigan, Mroz e*t al.* (1985) found no decrease in available P one and a half years after a whole tree clear cut.

The amounts of available K were significantly higher at the FS (148.15 mg kg^{-1}) than DS (84.14 mg kg^{-1}) (Table 1). Unlike P, which usually enters a soil through organic matter, K is most often found in the soil in inorganic forms, usually resulting from the mineral weathering of the rocks and parent material in the soil (Brady and Weil, 2008). Because it is often a mobile ion in soils, K losses to leaching can be significant (Alfaro *et al.*, 2004 a; Alfaro *et al.*, 2004 b; Khormali *et al.*, 2005). Clear cutting of the forest, DS, could have increased both lateral and vertical movement of water and resulted in leaching of K from the upper levels of the soil (Pennock and Kessel, 1997). Other studies report dramatic decreases in K following deforestation such as occurred in the unplanned and unregulated clear cut that resulted in the coppice in this study. In their whole tree clear cut study, Mroz *et al.* (1985) found K to decrease on three sites, from 1483 kg ha^{-1} to 508 kg ha^{-1}, 1230 kg ha^{-1} to 347 kg ha^{-1}, and 1396 kg ha^{-1} to 282 kg ha^{-1} respectively, a year and a half after the cut. Bormann *et al.* (1968) recorded a seven-fold rise in K levels found in runoff in the Hubbard Brook watershed study following a clear cut and regeneration suppression. In their deforestation study in Iran, Hajabbasi *et al.* (1997) found K levels to be 0.25 meq L^{-1} in the natural forest and only 0.15 meq L^{-1} in DS under cultivation. As with P, Boyle *et al.* (1973) found no decrease in long term K levels following the whole tree clear cut, but speculated that steeper slopes or regeneration suppression could result in significant nutrient losses. Eden *et al.* (1991) recorded similar results to this study in regard to the conversion of forest to pasture. In their study, the conversion of evergreen forest in Roraima, Brazil only resulted in a slight decrease in soil K, from 0.11 meq per 100 g^{-1} to 0.0911 meq per 100 g. In contrast, Abbasi *et al.* (2007) observed lower values in available K, from 69.911 meq per 100 g^{-1} to 44.411 meq per 100 g^{-1}, when comparing natural forest and grassland.

The surface variogram confirmed the isotrophic condition for soil features in the FS (Figure 3) and DS (Figures 4). The semivariogram models (Figures 5 and 6) and some of the geostatistical parameters of soil chemical features are shown in Table 2. Following deforestation the variogram model for soil pH was changed from linear to spherical (Table 2; Figures 5 and 6), the spatial class was enhanced from weak to medium spatial dependency (Table 2) and the fractal dimension was reduced from 1.99 to 1.81 (Figure 7).

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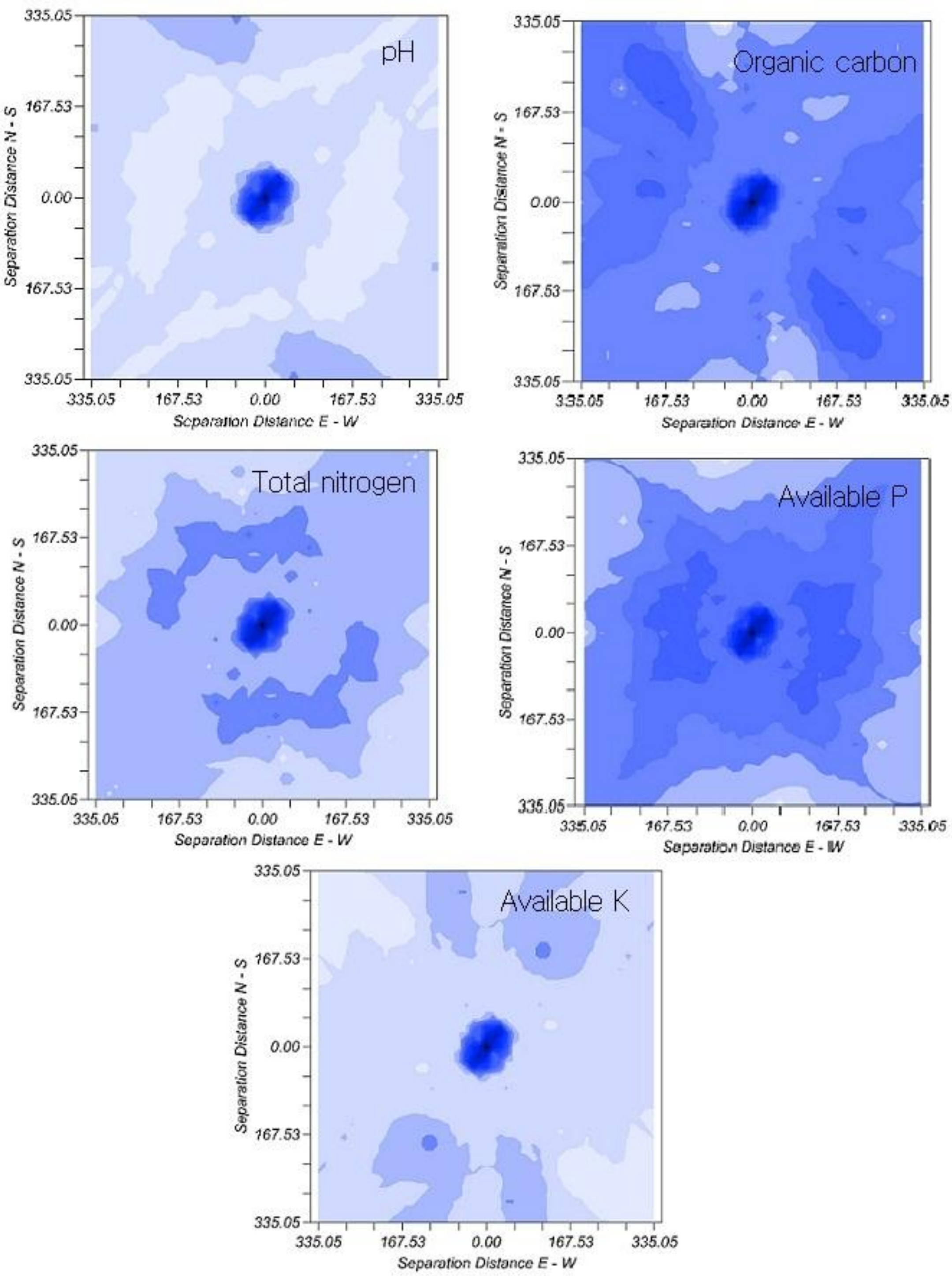

Figure 3 Surface variogram of soil features in the forest site (FS), lowland part of Khanikan forests located in Mazandaran Province, north of Iran

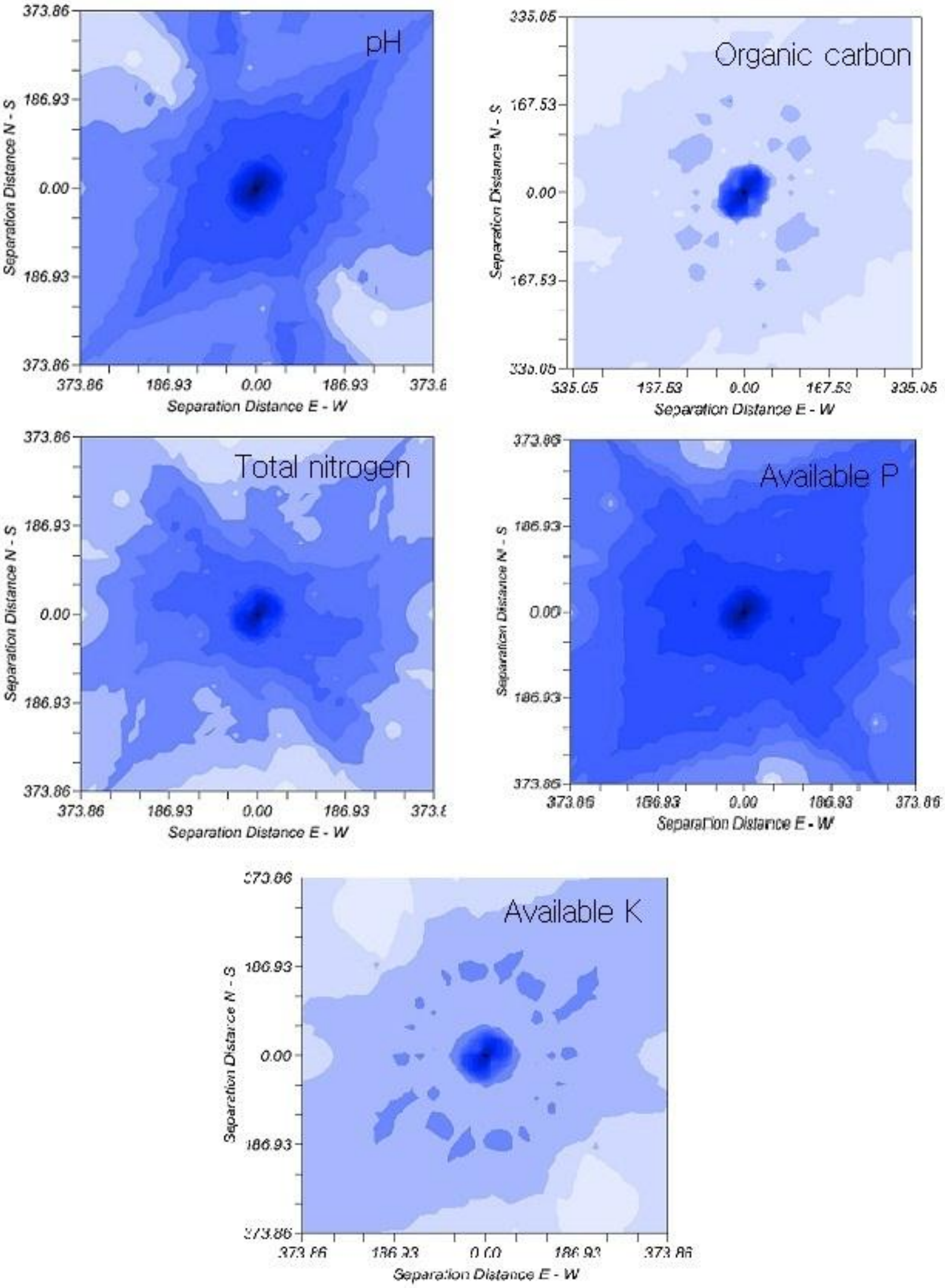

Figure 4 Surface variogram of soil features in the deforested site (DS), lowland part of Khanikan forests located in Mazandaran Province, north of Iran

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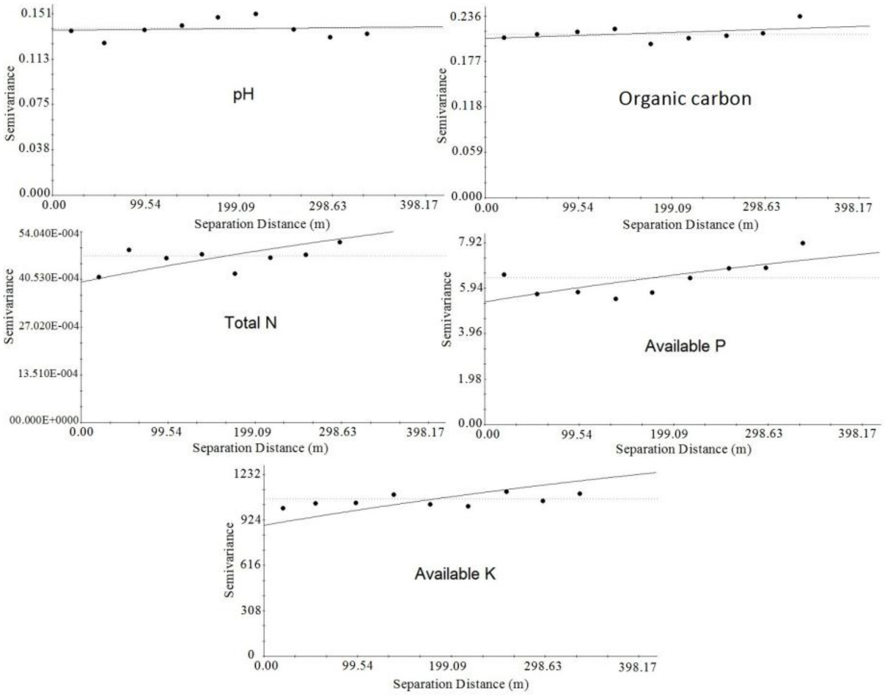

Figure 5 Semivariograms of soil features in the forest site (FS), lowland part of Khanikan forests located in Mazandaran Province, north of Iran

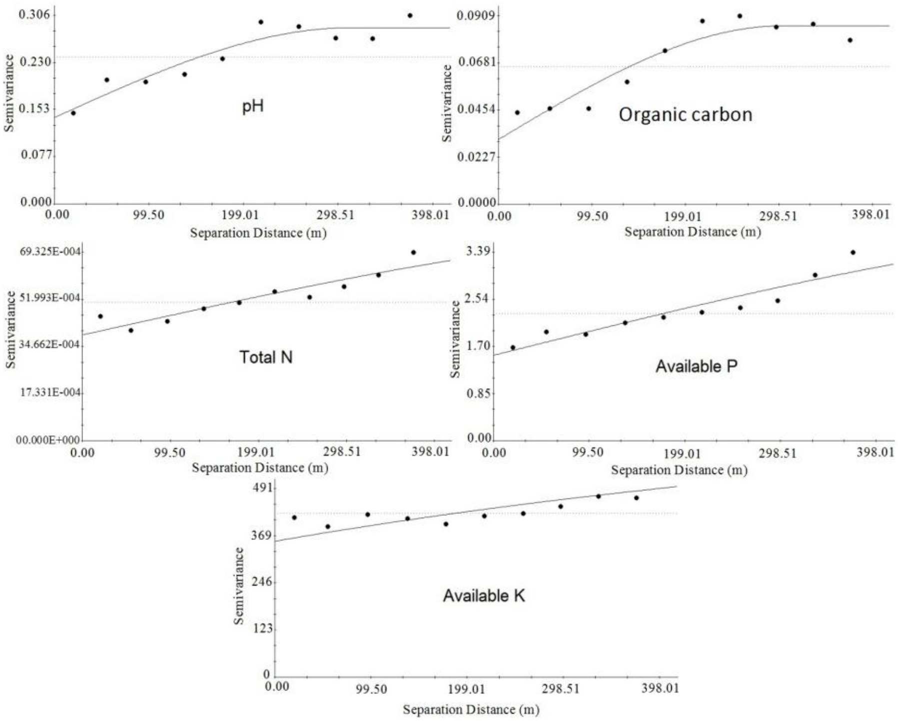

Figure 6 Semivariograms of soil features in the deforested site (DS), lowland part of Khanikan forests located in Mazandaran Province, north of Iran

Table 2 Parameters of semivariogram models for soil features in study forest (FS) and Deforested site (DS)

Variable	pH		Organic carbon (%)		Total N (%)		Available P (mg kg^{-1})		Available K (mg kg^{-1})	
	FS	**DS**	**FS**	**DS**	**FS**	**DS**	**FS**	**DS**	**FS**	**DS**
Model	L	S	L	S	E	S	E	S	E	E
Spatial part (%)	2.158	51.228	5.936	63.953	57.142	57.142	0.500	60.253	50.028	50.007
Spatial class	W	M	W	M	M	M	M	M	M	M
MBE	-	0.003	-	-0.000	-0.000	0.000	0.013	-0.001	0.493	0.087
MAE	-	0.301	-	0.174	0.056	0.054	2.057	1.162	28.683	18.114

Models: L = Linear; S = Spherical; E = Exponential, Spatial part given by the ratio (sill-nugget effect/sill) × 100, Spatial class: W = Weak spatial dependency; M = Medium spatial dependency, MBE = Mean Bias Error and MAE = Mean Absolute Error

Quantifying the spatial relationships among soil features and other component of ecosystems is essential to regional efforts for planning habitat preservation, zoning and land management and predicting the effects of deforestation from local to global ecosystems. The fundamental transformation of spatial pattern, from linear or exponential model in the FS to spherical form in the DS of soil features due to forest land degradation, is unquestionably related to the drastic removal of vegetation and its unavoidable consequences. In our study, most of soil characters are spatially independent with higher fractal dimension at the studied scale of FS. This is due to the heterogenized character of this ecosystem, caused by biotic factors.

In FS, different factors can be effective on variability of soil features such as different tree species, canopy gaps, log, snag, uprooted trees, etc. Thus, the variogram model tended to be linear or pur nugget effect form. But, when the vegetation is substantially removed, DS, the effective contribution of biotic factors to soil variables and their spatial variability is diminished, thus the soil features will be homogenized and the soil variables can be depended in more distance with lower fractal dimension (tending to exponential or spherical variogram model). The spherical models indicate distinct patches of large (or small) concentrations in a matrix of less (or greater) concentrations (Yavitt *et al.*, 2009). The observation that several models provide the best fit in our data indicates different spatial patterns among the set of soil feature. According to our finding, the spatial variability of soil pH and SOC were more imposed by deforestation compared to the other soil characters that is correspond with Nael *et al.* (2004) results. In FS, a linear model was fitted to the semivariograms of pH and SOC (Table 2).

This model indicates randomly distributed data pattern and suggests that the changes in semivariance (γ) with increasing lag distance are not significant and the total variance is found at all scales of sampling. In other words, there is no spatial dependence in the data points (Nael *et al.*, 2004). For the DS, a spherical model provided a significant fit to semivariograms of pH and SOC (Table 2 and Figure 6). This model suggests that these variables are spatially patterned and that the semivariance (γ) first rises and then levels off at the sill, indicating the distance beyond which samples are independent. Other features of this model are range and nugget; the former indicates the range over which samples show spatial dependence and the latter is the variance that exists at scales finer than the field sampling which is found at zero lag distance. The maps obtained by the kriging method (Figure 8 and 9) showed that the spatial distribution of the soil nutrients showed a direct relation with the increasing of soil pH (low acidity).

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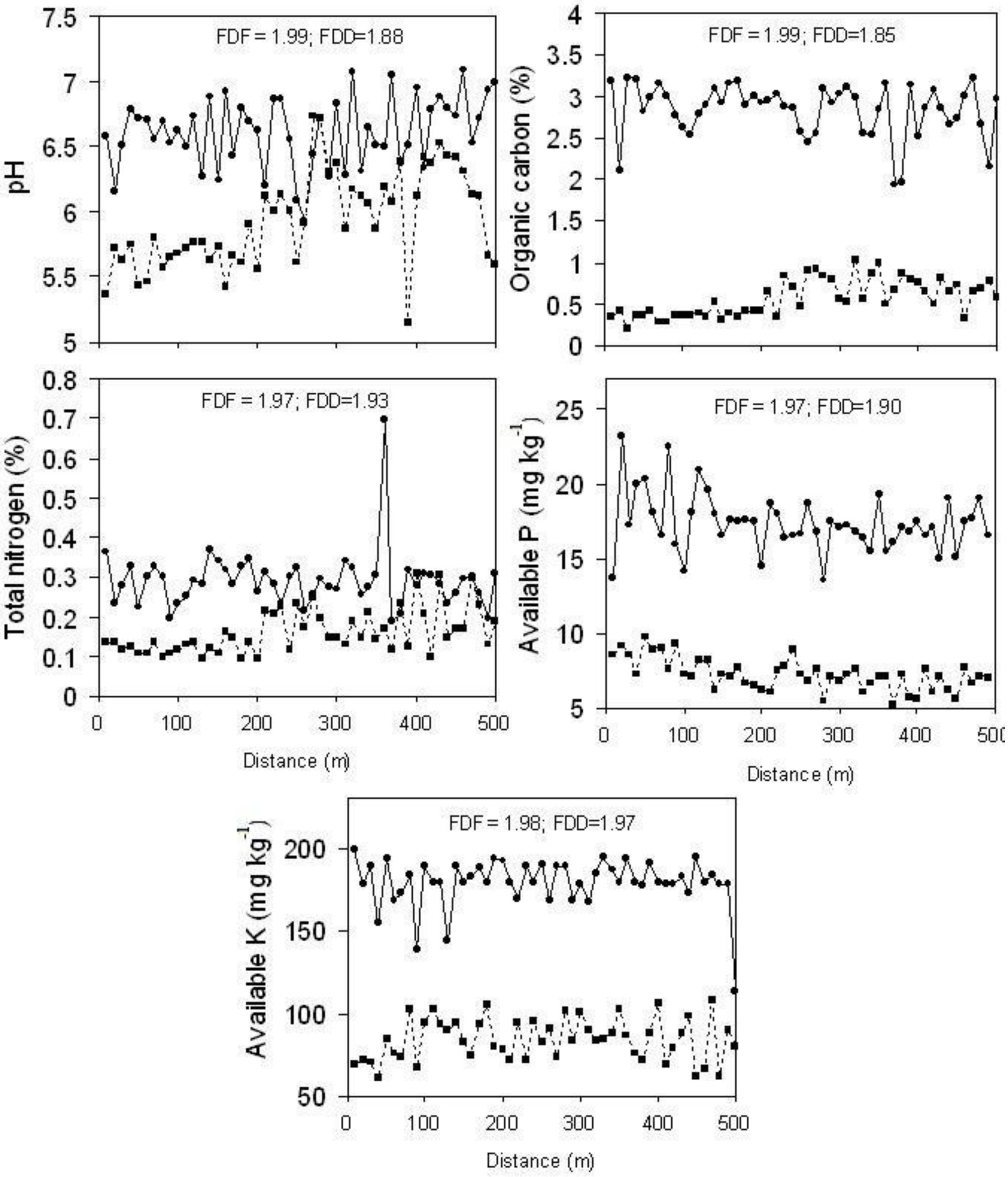

Figure 7 Variability of soil characters along the sampling line in study site. Solid circle with continues line represent values on every sampling of forest site (FC). Solid square with discrete line represent values on every sampling of deforested site (DS). FDF means Fractal Dimension of intact Forets. FDD means Fractal Dimension of Deforested site

As the same, after forest degradation the variogram model for SOC was changed from linear to spherical (Table 2; Figures 5 and 6), the spatial class was enhanced from weak to medium spatial dependency (Table 2) and the fractal dimension was reduced from 1.99 to 1.85 (Figure 7). About total N, the variogram model was changed from exponential to spherical (Table 2; Figures 5 and 6), the fractal dimension was reduced from 1.97 to 1.93 (Figure 7) and the spatial class was non-change as medium spatial dependency (Table 2) after occurrence of deforestation. The variogram model for soil available P was changed from exponential to spherical (Table 2; Figures 5 and 6), the fractal dimension was reduced from 1.97 to 1.90 (Figure 7) and the spatial class was non-change as medium spatial dependency (Table 2) after forest degradation.

The spatial varaiability of soil available K was less imposed by deforestation as the variogram model was detected as exponential and the spatial class was found as medium spatial dependency for both of FS and DS (Table 2; Figures 5 and 6). Also, the fractal dimension was a little reduced from 1.98 to 1.97 (Figure 7). A small, effective range implies a distribution pattern composed of small patches. In this study, the range of spatial dependence for soil chemical features in the 0-30 cm depth of DS was smaller than FS that are indicating a small patched distribution pattern (Table 2). Regarding to non-linear of soil features in FS (except for pH and SOC properties) and DS the contour maps of soil features prepared by ordinary kriging (Figures 8 and 9). The cross-validation results (Figures 10 and 11) of MBE and MAE are presented in Table 2.

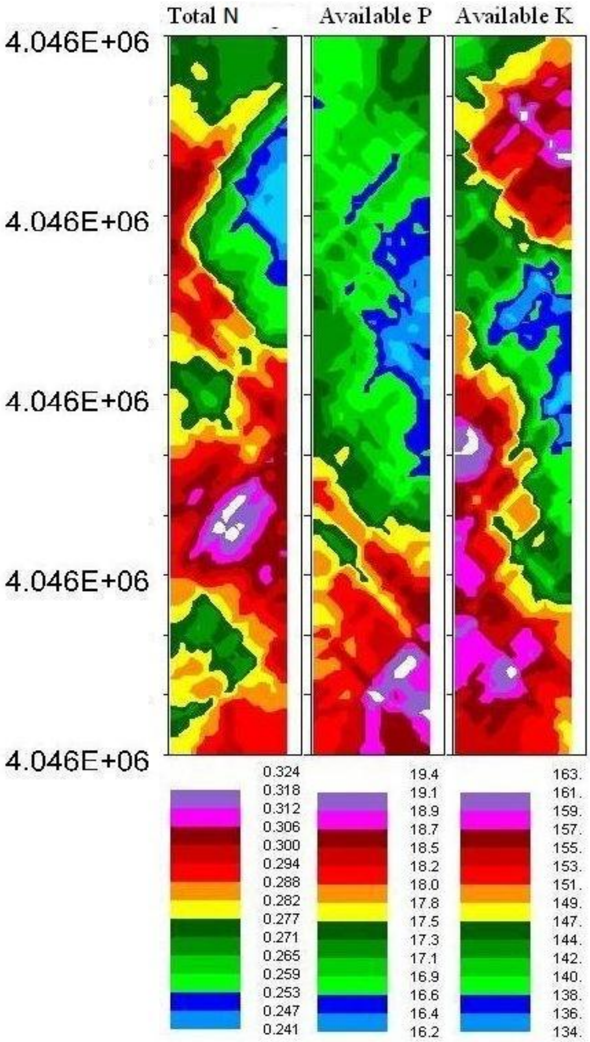

Figure 8 Contour maps of soil features prepared by ordinary kriging in the forest site (FS), lowland part of Khanikan forests located in Mazandaran Province, north of Iran

[DOR: 20.1001.1.23222700.2014.2.4.4.7]

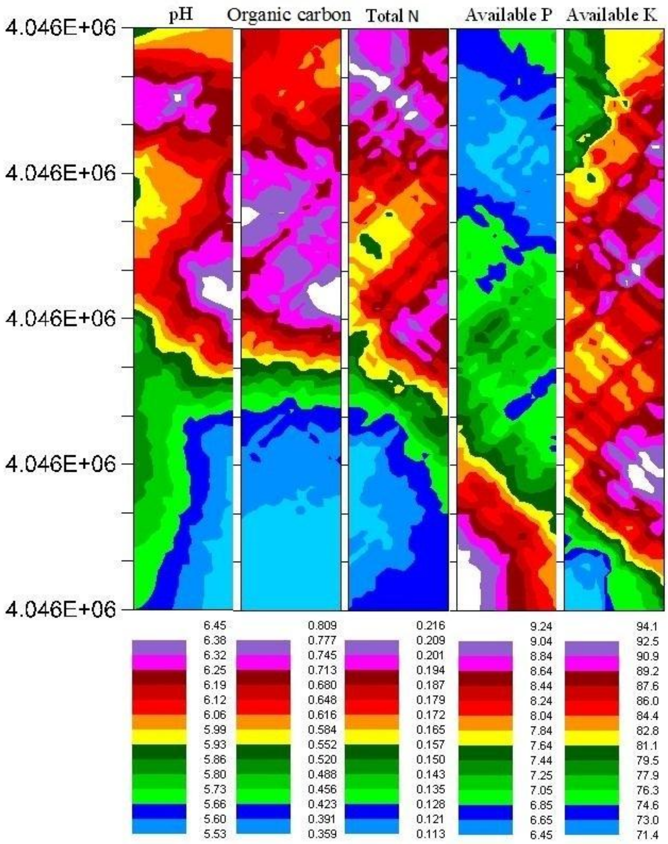

Figure 9 Contour maps of soil features prepared by ordinary kriging in the deforested site (DS), lowland part of Khanikan forests located in Mazandaran Province, north of Iran

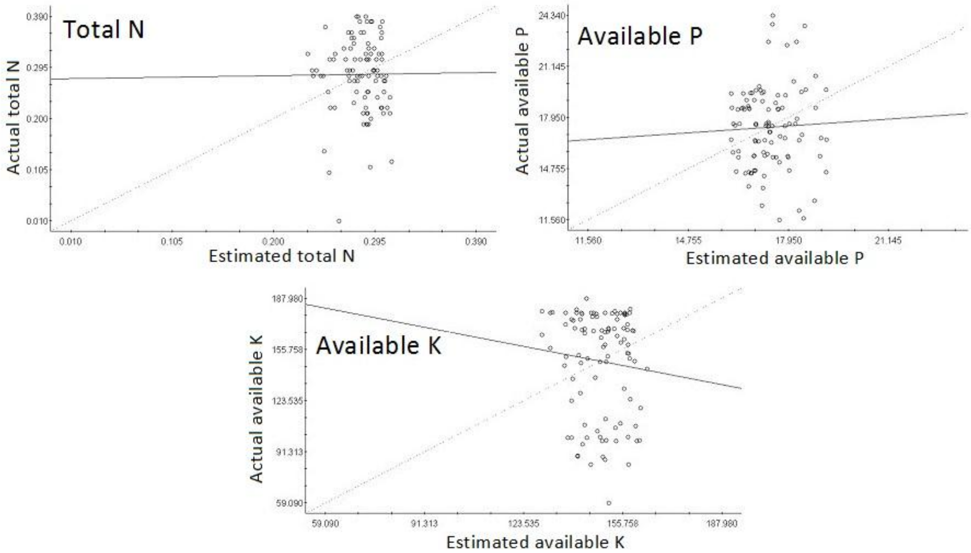

Figure 10 Comparison between measured and estimated values for soil features in the forest site (FS), lowland part of Khanikan forests located in Mazandaran Province, north of Iran

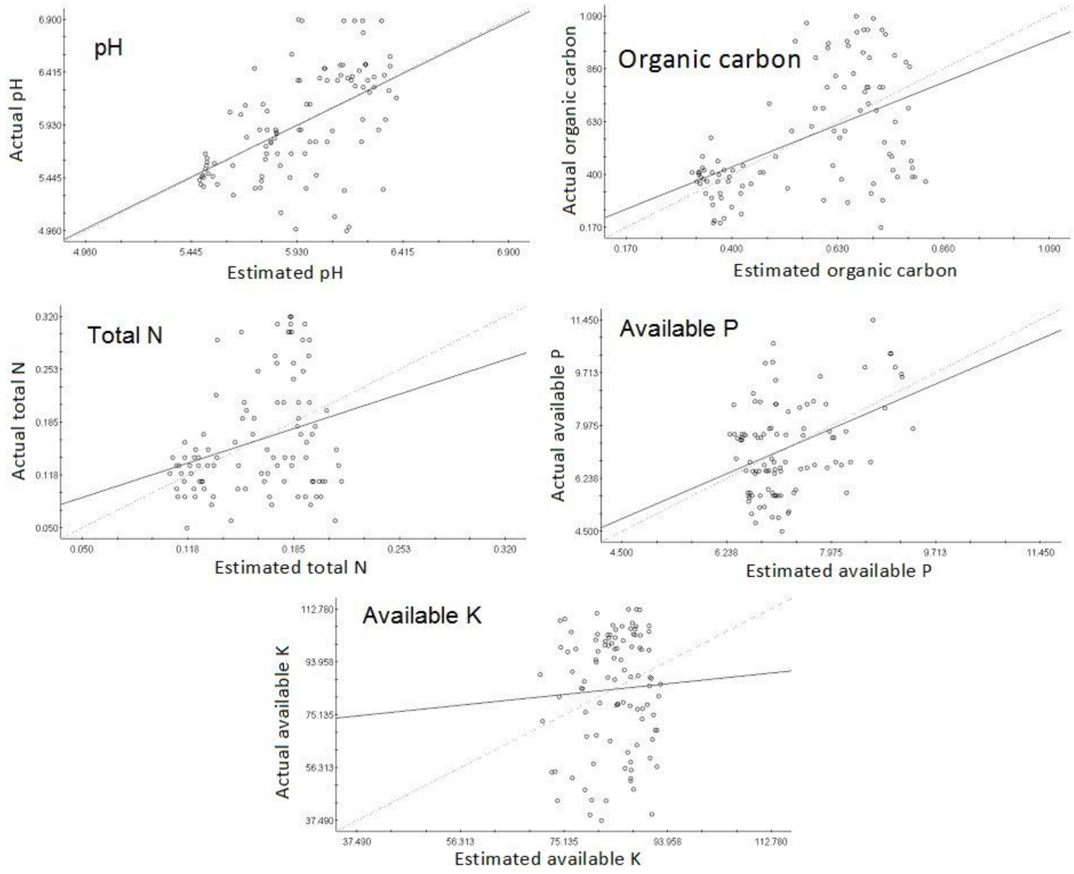

Figure 11 Comparison between measured and estimated values for soil features in the deforested site (DS), lowland part of Khanikan forests located in Mazandaran Province, north of Iran

4 CONCLUSION

In the current study, geostatistical method was used to investigate the spatial heterogeneity of soil nutrients features under an undisturbed forest site (FS) and a completely deforested site (DS) in lowland part of Khanikan forests located in Mazandaran Province, north of Iran. The results indicate that the soil nutrients features in the study site were moderately variable on the local scale. Deforestation was followed by the decreasing of soil reaction, organic carbon, total nitrogen, available phosphourous and potassium. A geostatistical analysis revealed that the deforestation changed the spatial variability models and fractal dimension of soil features. The spatial variability of soil pH and organic carbon content were more imposed by deforestation compared to the other soil features. Our results suggest that deforestation should be regarded as an effective factor on variability of soil chemical that are tied to forest ecosystem management.

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نقش تخریب جنگل بر تغییرپذیری مکانی عناصر غذایی خاک در یک جنگل هیرکانی

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چکیده: مطالعه تخریب جنگل از طریق ارزیابی عناصر غذایی خاک، بیان‌گر عملکرد خاک در اکوسیستم، به‌منظور مدیریت پایدار منابع سرزمین ضروری می‌باشد. تحقیق حاضر برای درک بهتر تغییرات عناصر غذایی خاک در نتیجه مدیریت بهره‌برداری با استفاده از برخی مشخصه‌های خاک و الگوی مکانی آنها صورت پذیرفت. در اواسط تابستان ۱۳۹۳ دو رویشگاه جنگلی (منطقه دست‌نخورده و منطقه کاملاً تخریب‌یافته) در بخش پایین‌بند جنگل‌های خانیکان چالوس، واقع در استان مازندارن- شمال ایران، مدنظر قرار گرفتند. در هر یک از مناطق فوق‌الذکر، تعداد ۱۰۰ نمونه خاک تا عمق ۳۰ سانتی‌متری در طول دو ترانسکت ۵۰۰ متری برداشت شد. فاصله ترانسکت‌های هر منطقه و هم‌چنین فاصله بین نمونه‌های خاک در طول هر ترانسکت ۱۰ متر در نظر گرفته شد. میانگین مقادیر pH خاک در منطقه تخریب‌یافته (۵/۷۰) کم‌تر از منطقه جنگلی (۶/۵۸) به‌دست آمد. میانگین مقادیر کربن آلی خاک در منطقه جنگلی (۲/۷۸ درصد) به‌طور معنی‌دار بیش‌تر از منطقه تخریب‌یافته (۰/۵۶ درصد) بوده است. به‌طور مشابه، نیتروژن خاک نیز به‌طور معنی‌دار در منطقه جنگلی (۰/۲۸ درصد) بیش‌تر از منطقه تخریب‌یافته (۰/۱۶ درصد) به‌دست آمد. میانگین مقادیر فسفر و پتاسیم خاک در منطقه جنگلی (۱۷/۳۳ و ۱۴۸/۱۵ میلی‌گرم بر کیلوگرم) تفاوت آماری معنی‌داری را با منطقه تخریب‌یافته (۷/۲۴ و ۸۴/۱۴ میلی‌گرم بر کیلوگرم) نشان داد. تجزیه و تحلیل زمین‌آماری بیانگر آنست که تخریب جنگل، مدل‌های تغییرپذیری مکانی و ابعاد فراکتالی مشخصه‌های خاک را تغییر داده است. به عنوان نتیجه، تغییرپذیری مکانی مشخصه‌های کربن آلی و pH خاک بیشتر از سایر مشخصه‌های مورد بررسی تحت تاثیر تخریب جنگل قرار گرفت. نتایج این پژوهش پیشنهاد می‌کند که تخریب جنگل باید به عنوان یک فاکتور موثر بر تغییرپذیری عناصر غذیی خاک مورد توجه قرار گیرد و این موضوع با مدیریت اکوسیستم جنگل در ارتباط می‌باشد.

کلمات کلیدی: بعد فراکتالی، تخریب جنگل، زمین آمار، شیمی خاک

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