Land-use changes and practical application of the land degradation neutrality (LDN) indicators: a case study in the subalpine forest ecosystems, Republic of Korea

Sangsub Cha, Chan-Beom Kim, Jeonghwan Kim, Ah Lim Lee, Ki-Hyung Park, Namin Koo and Yong Suk Kim

Forest Restoration and Resource Management Division, National Institute of Forest Science, Seoul, Republic of Korea

ABSTRACT

Recent estimates show that land degradation is rapidly progressing worldwide. Therefore, efforts are being made in Korea toward Land Degradation Neutrality (LDN) to extend the development of green areas. As part of this effort, artificial ecosystems created through land-use changes are restored to forests. We examined grassland created through land-use change, an afforested site planted for forest restoration, and a nearby primary forest according to the LDN indicators. The grassland created about 40 years ago showed higher bulk density and available phosphorus compared to the forest and showed relatively low carbon and nitrogen contents. According to the assessment of LDN indicators, the soil organic carbon stock and productivity calculated from the normalized difference vegetation index of the afforested site for restoring the grassland to forest did not change. The assessment of the restoration effect was not made in the short-term, but in near future, the planted trees are expected to grow, restoring the land to forest. This study shows that the LDN program of Korea should follow carefully established restoration policies and strategies to achieve positive results.

1. Introduction

Land resources and their preservation are garnering increasing concern worldwide. In 2011, the United Nations Food and Agriculture Organization (FAO) estimated that 33% of the world’s land and water resources are highly or moderately degraded (Food and Agriculture Organization of the United Nations 2011). Globally, land degradation has caused extensive damage, leading to significant economic losses (Pimentel et al. 1995; Balmford et al. 2008). Land degradation is the result of poor management of land resources. In particular, indiscriminate land-use change accelerates land degradation (Lambin et al. 2001; Hill et al. 2008; Kassa et al. 2017; Khaleedian et al. 2017).

Land-use change refers to the establishment of new vegetation and soil management, including changes in nutrient inputs, removal of plant biomass, and modification of soil structure (Poeplau and Don 2013). Land-use change, primarily to create agricultural cropland and pasture, has been a common human activity for centuries (Dupouey et al. 2002). Land-use changes have a significant impact on the global carbon (C) cycle due to changes in soil C stock rate and turnover, soil erosion, and vegetation biomass (Post and Kwon 2000; DeGryze et al. 2004; Laganière et al. 2010; Li et al. 2012). Forest and grassland are important

KEYWORDS

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CONTACT Yong Suk Kim sosilys@korea.kr Forest Restoration and Resource Management Division, National Institute of Forest Science, Seoul 02455, Republic of Korea

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productivity reflects relatively rapid changes; and SOC reflects slower changes that suggest the trajectory and proximity to thresholds, and can be an important indicator of overall soil quality (United Nations Convention to Combat Desertification 2016).

Before beginning the monitoring of indicators of LDN in Korea, we evaluated three adjacent land-use types within the same environment according to LDN indicators to apply them to the Korean ecosystem with the sub-national scale. This study was carried out in primary deciduous forest, a grassland created by altering the land cover of forest about 40 years ago, and an afforestation area planted 13 months ago to restore the grassland into a forest.

Our investigation addressed three main research questions:

1. How did soil properties respond to changes in land type due to land-use changes?
2. When assessed using the LDN indicators, did the three land types differ?
3. What was the result of the LDN assessment in the recently afforested area in terms of restoration?

We expect that the results of this study and the answers to the research questions will provide a basis for assessments of the impacts of land-use changes and LDN in temperate deciduous forests in Korea.

2. Materials and methods

2.1. Site description

This study was conducted near Maebong (37°45’39” N, 128°42’41” E; elevation 1,173 m above sea level) in Daegwanryong, Gangwon Province, Republic of Korea.

The soil is a brown forest soil (Inceptisol, USDA Soil Taxonomy) originating from granite. This area is covered with primary forest composed mainly of deciduous broadleaf trees, dominated by trees in the genus Quercus, such as Quercus mongolica and Quercus serrata. About 40 years ago, parts of the forest were reclaimed as grassland and used to raise livestock. This grassland covered an area of about 2000 ha, mainly with Poa pratensis, Festuca ovina, Artemisia princeps, and Rumex acetosella. After a recent decline in the number of grazing livestock, trees were planted to restore some of the grassland to forest. In September 2017, about 5 ha of grassland was planted with saplings at a density of 3000–4000 stems/ha, including Pinus densiflora, Abies holophylla, and Abies nephrolepis. The afforestation area was located adjacent to grassland and forest areas (Figure 1). We selected an area of about 1.5 ha as an experimental site within each of the three land-use types.

The average annual temperature and annual precipitation from 1980 to 2010 at the Daegwallyeong meteorological station, the nearest station to the experimental site, were 6.6°C and 1898 mm, respectively. The maximum temperature of 15.7–19.1°C occurred in summer (June through August), and precipitation was concentrated in summer, with monthly precipitation in the range of 201–327 mm. During winter (December through February), the average monthly temperature was −5.5 to 4.4°C and monthly precipitation was 37–54 mm.

2.2. NDVI data acquisition

Image acquisition of the experimental site was carried out using a multispectral camera (RedEdge-MTM; MicaSense, Seattle, WA, USA) on a custom-built
unmanned aerial vehicle. Spectral bands of blue (475 nm), green (560 nm), red (668 nm), near infrared (840 nm), and red edge (717 nm) were measured with a resolution of 1280 \times 960 pixels from a height of 50 m. The ground sample distance of the captured data was 16.61 cm, and data for a total area of 142 ha was obtained. Data collection took place in August 2018 on a clear day.

Pix4DMapperPro desktop software (Pix4D S.A., Lausanne, Switzerland; http://pix4d.com) was used to generate an orthomosaic map for each spectral band. After extracting the spectral band value of each image and calculating the normalized difference vegetation index (NDVI), a map of NDVI values was created. NDVI was calculated as follows (Pettorelli et al. 2005):

\[
\text{NDVI} = \frac{(\text{NIR} - \text{RED})}{(\text{NIR} + \text{RED})}
\]

where NIR and RED are the amounts of near-infrared and red light, respectively.

Plot-level NDVI values were extracted using ArcGIS ver. 10.4 (Esri, Redlands, CA, USA). The average NDVI value of each plot was obtained by dividing each experimental site into eight plots. One plot at an experimental site consisted of about 45,000 data points.

The equation from Bai et al. (2008) uses the annual sum NDVI value to calculate net primary productivity (NPP); the NDVI data in our study were not suitable for such calculation, but it was used to compare the NPP values of the three land types. NPP was calculated as follows (Bai et al. 2008):

\[
\text{NPP} [\text{kgCha}^{-1}] = 1106.37 \times \text{NDVI} - 564.55
\]

2.4. Statistical analysis

Differences among land-use types in measured soil physical and chemical properties were analyzed with one-way analysis of variance (ANOVA) followed by the Shapiro–Wilk test for normality, Levene’s test for homogeneity, and Duncan’s new multiple range test (p < 0.05). Principal component analysis (PCA) was also carried out to identify the major factors responsible for variations in the soil sample characteristics. To carry out PCA, the values of the soil properties analyzed were standardized as follows:

\[
\text{Zij} = \frac{(\text{Xij} - \text{X})}{\text{SD}}
\]

where Zij is the value of the standardized variables, Xij is the value of the variable for the sample of soil i at land use type j, and X and SD are the mean value and standard deviation, respectively. All statistical analyses were conducted using SPSS ver. 23.0 (IBM Corp., Armonk, NY, USA).

3. Results

3.1. NDVI by land-use type

Figure 2 shows the NDVI map for July 2018 in the experimental region. On the map, the forest region on the right side and the grassland on the left side are clearly distinguished. The NDVI values for each land-use type are shown in Table 1. The NDVI values of the grassland (0.823) and afforestation (0.821) sites did not differ significantly, while forest had an NDVI of 0.928, higher than those of the other two sites. The calculated productivities of the forest, grassland, and afforestation sites based on the NDVI were 462.98, 346.01, and 343.30 kg C/ha, respectively, and the productivities of the grassland and afforestation site were 25.3 and 25.9% lower than that of the forest site, respectively.

3.2. Soil properties according to land-use type

Table 2 shows the soil physical and chemical properties of the three land-use types. The C concentration, soil bulk density, SOC stock, TN, pH, CEC, AP, and exchangeable nutrients such as Ca\(^{2+}\) and K\(^+\) for all sites with the three vegetation types are shown as box plots in Figure 3.
C concentrations differed significantly between the forest and other two groups in the 0–10 cm soil layer. Grassland and afforestation sites had average C concentrations of 69.33 and 66.21 g/kg, respectively, whereas forest sites had an average of 90.06 g/kg (Table 2). The 10–20 cm soil layer showed significant differences among the three land types. The forest, grassland, and afforestation sites had C concentrations of 60.24, 40.75, and 45.82 g/kg, respectively.

On the other hand, the soil bulk density showed the opposite tendency as C concentration. Soil bulk density was higher in the grassland and afforestation sites than in the forest site at all soil depths. The bulk densities of the 0–10 cm soil layer were 0.607, 0.823, and 0.869 g/cm³ in the forest, grassland, and afforestation sites. The 10–20 cm soil layer had bulk densities of 0.775, 0.902, and 0.910 g/cm³ at the three land types, respectively.

Soil C stocks showed differing tendencies depending on soil depth. The 0–10 cm soil layer showed no significant differences in soil C stocks between the forest (53.51 ton/ha), grassland (56.76 ton/ha), and afforestation (55.43 ton/ha) sites, whereas in the 10–20 cm layer, the amount of stored C decreased from forest (46.46 ton/ha) to the afforestation (36.75 ton/ha) and grassland (41.20 ton/ha) sites. When the C stock of the 0–20 cm soil layer was calculated, there was no significant difference among the forest (99.08 ton/ha), grassland (93.52 ton/ha), and afforestation (96.64 ton/ha) sites. TN showed the same tendency as C in the 0–10 and 10–20 cm soil samples.

Soil pH and AP were higher in the grassland and afforestation sites than in forest in the 0–10 cm layer. CEC and Ca²⁺ showed no significant differences among the three sites. In the 10–20 cm layer, pH, AP, and Ca²⁺ were high in the afforestation site or grassland and afforestation sites, whereas K⁺ showed no significant difference among the three land types. CEC was highest in forest and lower in the grassland and afforestation sites.

### 3.3. Major factors affecting the differences

PCA was used to analyze ten soil properties monitored in each experimental site using the entire dataset from the soil samples collected (Table 3). The three main factors for the 0–10 and 10–20 cm soil layers accounted for 84.1 and 87.4% of the total variance, respectively; the loadings of the various soil properties considered for each factor are listed in Table 3. In the 0–10 cm soil samples, factor 1, accounting for 37.8% of the total variance, showed high loadings for TN, C, CEC, and K⁺. Similarly, factor 1 for the 10–20 cm layer showed high loadings for C, TN, CEC, C stock,
and soil bulk density. Factor 2 related to pH and Ca\(^{2+}\) for both soil layers. Factor 3 included AP and C stock for the 0–10 cm soil layer, and AP and K\(^{+}\) for the 10–20 cm soil layer. The distribution of the soil samples on the plane defined by factors 1 and 2, which explained 66.7% (0–10 cm layer) and 67.2%...
(10–20 cm layer) of the total variance, could be separated into clusters corresponding to forest and grassland sites (Figure 4). Soil samples from the afforestation site were difficult to classify into clusters due to their wide distribution.

4. Discussion

4.1. Differences in NDVI according to vegetation type

The purpose of this study was to compare soil in different land types according to land-use changes and to evaluate these land-use types according to LDN indicators. Therefore, we compared three adjacent areas: broadleaf deciduous forest, grassland that was created 40 years prior to sampling by altering the land cover of a forest, and an afforestation zone that was planted to restore grassland to forest about 1 year prior.

Our results revealed changes in vegetation caused by land-use changes through a remote sensing method using the NDVI. The NDVI values of different land-use types, especially forest and grassland, were clearly distinguished. Although the NDVI differed depending on the timing of data collection and weather conditions, the NDVI value of the forest site was very high, about 0.93, and the grassland site showed a difference of 0.1 or more, with a value of about 0.82. The NDVI values of the grassland and afforestation sites were similar. The afforestation site was planted in September 2017, and its NDVI did not differ markedly from that of the grassland, as only about 1 year had passed since planting. The distinct differences in NDVI values of the forest, grassland, and afforestation sites suggest that the NDVI is suitable for classifying vegetation type differences. Because the NDVI provides information about the distribution of vegetation communities (Reed et al. 1994), vegetation biomass (Reed et al. 1994), and the extent of land degradation in various ecosystems (Holm et al. 2003; Thiam 2003), it can be used for a variety of purposes in ecological research.

In general, NDVI values differ strongly depending on the timing and weather conditions when the data are collected. Therefore, it is necessary to use time-series data to obtain valid results without noise (Pettorelli et al. 2005). Unlike previous studies based on satellite imagery, the equipment used in this experiment collected measurements at a height of 50 m or less from the ground. This method should be helpful for constructing a stable dataset because the influence of the surrounding environment is relatively small.

The relationship between NDVI and vegetation productivity has been well documented. The productivity of

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**Table 3.** Loadings of the factors identified through PCA based on ten soil properties of the forest, grassland, and afforestation sites.

|          | 10 cm Factor I | Factor II | Factor III | 20 cm Factor I | Factor II | Factor III |
|----------|----------------|-----------|------------|----------------|-----------|------------|
| Organic C | 0.961          | 0.067     | −0.196     | Organic C      | 0.980     | −0.045     |
| Total N  | 0.949          | −0.018    | −0.173     | Total N        | 0.948     | 0.023      |
| CEC      | 0.893          | 0.254     | 0.032      | C stock        | 0.890     | 0.218      |
| C stock  | 0.698          | 0.227     | 0.557      | C stock        | 0.851     | 0.067      |
| pH       | −0.096         | 0.92      | 0.177      | Bulk density   | −0.645    | 0.447      |
| K⁺       | 0.570          | −0.375    | 0.177      | Available P    | −0.037    | 0.950      |
| Ca²⁺     | 0.323          | 0.888     | 0.226      | Ca²⁺           | 0.215     | 0.153      |
| Available P | −0.029    | 0.231     | 0.867      | pH             | −0.179    | −0.181     |
| Bulk density | −0.665    | 0.069     | 0.693      |                |           |            |

**Figure 4.** (A and B) Distribution of soil samples in the plane defined by factors 1 and 2 from the PCA performed on the main properties of soil samples from the forest (open circles), grassland (gray circles), and afforestation (filled circles) sites.
the three sites varied based on the relationship model between NDVI and NPP described by Bai et al. (2008), with productivity at the forest site about 25% higher than that of the other two land types. However, because this model equation was derived using MODIS NPP with a 1 km resolution, there may be constraints on its reliability due to the limited information available on land use and species composition when the vegetation cover per unit area is very complex. Generally, forest shows higher productivity than grassland. According to Knapp and Smith (2001), the aboveground NPP of several biomes showed a positive correlation with precipitation, while mixed deciduous forest, including Harvard Forest and Hubbard Brook, in biomes with similar precipitation had productivities 3.5–3.7 times higher than that of Niwot Ridge, with moist alpine meadows. Therefore, this study may have underestimated the difference in productivity between the forest, grassland, and afforestation sites.

4.2. Changes in soil properties

Changes in vegetation type due to land-use changes have a major impact on ecosystem functions and soil characteristics (Poeplau and Don 2013). Several studies have investigated soil property changes due to land-use changes, and have found that soil C storage is a particularly major concern (Guo and Gifford 2002; Don et al. 2011). In our study, changes in soil properties due to land-use changes were also observed.

When land is restored from grassland to forest, several different tendencies of change in the soil C stock have been reported. Don et al. (2011) reported a soil C stock increase during the change from grassland to secondary forest, while Guo and Gifford (2002) reported a decrease in soil C stock when shifting from grassland to secondary forest based on meta-analysis. In our results, the C concentration was highest in the forest soil, but the soil C stock, despite showing no statistically significant differences, was highest at the afforestation site. These results can be explained by differences in soil bulk density. Bulk density changes are important factors affecting soil C stock changes because C stocks depend linearly on both soil C concentration and bulk density (Don et al. 2011). Soil bulk density increased sharply with land-use change from primary forest to grassland (Don et al. 2011). Soil compaction, a characteristic of grassland soil, is defined by an increase in mass per unit volume due to compression of soil aggregates and a decrease in pore volume and continuity compared with a well-structured soil (Batey 2009). Trampling by livestock during grazing and trafficking of machinery for grassland management are likely causes of soil compaction (Newell-Price et al. 2013). As the bulk density increases due to soil compaction, soils are sampled with the same volume, resulting in greater sampled soil mass. Because soil mass is ultimately associated with soil C, sampling a larger mass of soil yields higher SOC stocks (Davidson and Ackerman 1993).

Our results showed that grassland had similar soil C stocks, despite relatively low productivity, compared to the forest site. Generally, most C accumulation in grassland ecosystems occurs belowground (Soussana et al. 2004) due to high annual belowground productivity from root production of herbaceous plants. Conversely, forest ecosystems accumulate large amounts of C aboveground. According to Zimmermann et al. (2010), the ratio of belowground: aboveground C was 15.8 in grassland and 2.1 in forest, indicating that forest ecosystems accumulate C aboveground while grassland ecosystems accumulate C belowground. Although the soil C stock did not differ significantly among land-use types, when divided by soil depth, differences among land types were observed. Unlike the 0–10 cm soil layer, the maximum C stock in the 10–20 cm layer was observed in forest. Zimmermann et al. (2010) reported that the soil C stock of grassland soil decreased rapidly with depth, while forest soil was less affected by depth. Our results showed that C stocks in deep grassland soil were significantly reduced compared to forest soil. The ratio of 10–20 cm layer to 0–10 cm layer C stocks in forest soil and grassland soil were 0.89 and 0.68, respectively. Accordingly, we predicted that deep soil would contain more C in forest than grassland samples; therefore, the total C stock of forest soil would be higher than that of other land types. The afforestation site showed a high ratio of 0.8, but its coefficient of variation was about 3 times higher than that of the other soils. During the planting process, the soil layers were disturbed, and the upper soil may have been introduced into the lower soil layer.

Land-use change had some effect on the nutrient status of the soil. Although not all nutrients showed statistically significant differences, pH and available nutrients were high in soil from the grassland and afforestation sites, while C and TN concentrations showed the opposite trend. Highly productive ecosystems use nutrients for the production of plant biomass, while low-productivity ecosystems use microbial processes to store nutrients in soil (Grigulis et al. 2013). And soil compaction of grasslands can reportedly have negative implications for the ability of the soil to support important ecosystem services, such as productivity (Frost 1988; Newell-Price et al. 2013). In this study, the forest site, with high productivity, appeared to have low nutrient contents because available nutrients were absorbed from the soil and used in plant biomass production. Whereas, some available nutrient of soils, Ca and AP, showed a different pattern. In particular, AP concentrations in the grassland and afforestation site soil were 2.5 to 7.7 times higher than those of forest soil, this is thought to be due to the fertilization. This effect is also evident at pH.

To compare soil quality in samples from each land type, all soil properties should be considered simultaneously. PCA was used to investigate the relationships among variables and to identify the main factors that differentiate each soil sample. Forest samples, with higher values along Axis 1, were distinguished from
the grassland samples, as indicated by their tight clustering in Figure 4. These samples were mainly characterized by high levels of C, TN, CEC, and bulk density in the 0–10 cm soil layer. The C stock in the 0–20 cm soil layer was added as a primary characteristic to Axis 1. This result is in accordance with many previous studies showing that soil C changes are a major effect of land-use changes (Guo and Gifford 2002; Don et al. 2011).

4.3. Evaluation based on LDN indicators

We evaluated three land types according to LDN indicators based on the results of the present study. The LDN indicators are used to measure land degradation by evaluating changes in land cover, productivity, and soil organic C based on the UNCCD criteria (2016). A location is considered to be degraded if at least one of the three indicators shows a negative change. This criterion is known as the “one out, all out” rule (Cowie et al. 2018). In the present study, we compared forest and grassland site, grassland and afforestation site, respectively, in the order in which artificial interference was applied.

First, land cover is assessed as the vegetation type covering the land. Grassland have been maintained since it was converted by land use changes about 40 years ago. Land cover of grassland site has been maintained since it was converted by land use changes about 40 years ago. Convert from forest to grassland is assessed as degradation with “vegetation loss”. The afforestation site was planted about 1 year prior to the study in grassland, but the growth of the trees was not yet sufficient to create a forest; therefore, it was defined as shrubland. Shrubland is classified as a type of grassland according to the land cover classes in the System of Environmental-Economic Accounting, which follows the FAO Land Cover Meta Language. Similarly, the NDVI values in this experiment were similar for the grassland and afforestation sites. Thus, the land cover class did not change following restoration. However, the afforestation site is expected to develop into a forest after the planted trees grow for a few years.

Second, land productivity can be estimated using the total aboveground NPP, which is defined as the energy fixed by plants minus their respiration (United Nations Convention to Combat Desertification 2016). This indicator can be calculated over large areas from earth observation data. Vegetation indexes, most commonly the NDVI, provide realistic proxies for NPP that are used widely (Yengoh et al. 2014). In our results, the NDVI values of the forest differed from those of the grassland and afforestation sites, and productivity calculated from NDVI (Bai et al. 2008), differed by about 25%. The results for the afforestation site confirmed that it has not increased or decreased in productivity, as the grassland and afforestation site showed similar NDVI values.

Third, SOC is estimated from aboveground and belowground C stocks. SOC is an indicator of overall soil quality associated with nutrient status, water content, and aggregate stability and structure (United Nations Convention to Combat Desertification 2016). In this study, the SOC of the 0–20 cm soil layer did not differ significantly among the three land types. However, deeper soils at the forest site are expected to contain more C than grassland and afforestation site soils, whereas the aboveground C storage of forests is greater than that associated with other vegetation types (Zimmermann et al. 2010).

Based on the above results, the forest site, the baseline, maintains a tree-covered area and accumulates the largest amount of C among the three land types analyzed, and also has the highest productivity because the forest site has been maintained as a stable forest ecosystem. The grassland site, after undergoing land-use change, has been degraded in forest and maintained for about 40 years. On the other hand, evaluation of the afforestation site is complex. Although all the indicators did not show a difference due to the restoration. However, the belowground C stock in the afforestation site soil showed no significant difference, but was higher than that of the grassland soil; however, the aboveground C stock is expected to be greater than that of the grassland, because it has been planted with saplings at a density of 3000–4000 stems/ha. And, if the trees planted in the afforestation area grow, this assessment may change in the near future.

To summarize our evaluation of the three land types, the forest site had consistent tree cover, high productivity, and SOC, and remained stable over time. The grassland site evaluated as degraded according to the LDN indicator due to land-use change since about 40 years ago, but this place seems to have stabilized as grassland over a long period of time. The afforestation site was not different from the grassland, so it was evaluated as ineffective due to planting to restore the grassland to forest. However, in a few years, this site is expected to be evaluated as improving when the planted trees have grown and the forest cover has settled.

Through this study, we confirmed that afforestation to restore the grassland to forest could not be assessed in the short term. The LDN plan in Korea is part of an effort to mitigate the progress of land degradation worldwide. This result suggests that the LDN of Korea must establish long-term restoration policies and strategies to achieve positive results. We expected that the results of this study will provide useful information for establishing and evaluating policies regarding land-use changes and LDN in the future.

5. Conclusion

The present study demonstrated that the physical and chemical properties of soils changed according to land-use change in the northeastern region of Korea and showed that artificial interference could have a negative impact on the evaluation according to the LDN indicator. The grassland and afforestation site showed
higher bulk density, available phosphorus and pH compared to the forest, and showed relatively low carbon and nitrogen content. According to evaluate of LDN indicator, grassland was lower than that of forest in land cover and NPP. This difference is the result of land-use change, and it was evaluated as “degradation”, although it is an ecosystem that has been maintained for about 40 years. For afforestation site, although show a little difference, from the grassland in NPP and SOC, it was evaluated as “no change” because did not show significant difference from the grassland in NPP and SOC. Our results indicate that the effect of restoration may not be detected by LDN indicator in the short term, although conversion of grasslands into forests is “improving” in Korea. Therefore, we suggest that the restoration policies and strategies should be carefully established in order to achieve the positive results of LDN and that the evaluation criteria of the restoration process need to be clearly distinguished according to some circumstances.

**Disclosure statement**

No potential conflict of interest was reported by the authors.

**References**

Bai ZG, Dent DL, Olsson L, Schaepman ME. 2008. Proxy global assessment of land degradation. Soil Use Manag. 24(3): 223–234.

Balmford A, Rodrigues SL, Walpole M, Ten Brink P, Kettunen M, Braat L, Groot R. 2008. The economics of biodiversity and ecosystems: scoping the science. Cambridge, UK: European Commission.

Batey T. 2009. Soil compaction and soil management – a review. Soil Use Manag. 25(4):335–345.

Cookson WR, Osman M, Marschner P, Abaye DA, Clark I, Murphy DV, Stockdale EA, Watson CA. 2007. Controls on soil nitrogen cycling and microbial community composition across land use and incubation temperature. Soil Biol Biochem. 39(3):744–756.

Cowie AJ, Orr BJ, Castillo Sanchez VM, Chasek P, Crossman ND, Erlewein A, Louwage G, Maron M, Metternicht GL, Minelli S, et al. 2018. Land in balance: the scientific conceptual framework for land degradation neutrality. Environ Sci Policy. 79:25–35.

Davidson EA, Ackerman IL. 1993. Changes in soil carbon inventories following cultivation of previously untilled soils. Biogeochemistry. 20(3):161–193.

DeGryse S, Six J, Paustian K, Morris SJ, Paul EA, Merckx R. 2004. Soil organic carbon pool changes following land-use conversions. Glob Chang Biol. 10(7):1120–1132.

Don A, Schumacher J, Freibauer A. 2011. Impact of tropical land-use change on soil organic carbon stocks – a meta-analysis. Glob Chang Biol. 17(4):1658–1670.

Dupouey JL, Dambrine E, Lafitte JD, Moares C. 2002. Irreversible impact of past land use on past land use on forest soils and biodiversity. Ecology. 83(11):2978–2984.

Food and Agriculture Organization of the United Nations. 2011. The state of the world’s land and water resources for food and agriculture (SOLAW)-managing systems at risk. Food and Agriculture Organization of the United Nations, Rome and Earthscan, London, UK.

Frost JP. 1988. Effects on crop yields of machinery traffic and soil loosening Part 1. Effects on grass yield of traffic frequency and date of loosening. J Agr Eng Res. 39(4):301–312.

Grigulis K, Lavorel S, Krainer U, Legay N, Baxendale C, Dumont M, Kastl E, Arnoldi C, Bardgett R.D, Poly F, et al. 2013. Relative contributions of plant traits and soil microbial properties to mountain grassland ecosystem services. J Ecol. 101(1):47–57.

Guo LB, Gifford RM. 2002. Soil carbon stocks and land use change: a meta analysis. Global Change Biol. 8(4):345–360.

Hill J, Stellmes M, Udelhoven T, Röder A, Sommer S. 2008. Mediterranean desertification and land degradation: mapping related land use change syndromes based on satellite observations. Glob Planet Change. 64(3–4):146–157.

Holm AM, Crioldland SW, Roderick ML. 2003. The use of time-integrated NOAA NDVI data and rainfall to assess landscape degradation in the arid shrubland of Western Australia. Remote Sens Environ. 85(2):145–158.

Kassa H, Dondeyne S, Poesen J, Frankl A, Nyssen J. 2017. Transition from forest-based to cereal-based agricultural systems: a review of the drivers of land use change and degradation in Southwest Ethiopia. Land Degrad Develop. 28(2):431–449.

Khaledian Y, Kiani F, Ebrahimi S, Brevik EC, Atkhenhead-Peterson J. 2017. Assessment and monitoring of soil degradation during land use change using multivariate analysis. Land Degrad Develop. 28(1):128–141.

Knapp AK, Smith MD. 2001. Variation among biomes in temporal dynamics of aboveground primary production. Science. 291(5503):481–484.

Kist G, Andreeva O, Cowie A. 2017. Land degradation neutrality: concept development, practical applications and assessment. J Environ Manage. 195:16–24.

Laganère J, Paré D, Bradley R.L. 2010. How does a tree species influence litter decomposition? Separating the relative contribution of litter quality, litter mixing, and forest floor conditions. Can J For Res. 40(3):465–475.

Lambin EF, Turner BL, Geist HJ, Agbola SB, Angelsen A, Bruce JW, Coomes OT, Dirzo R, Fischer G, Folke C, et al. 2001. The causes of land-use and land-cover change: moving beyond the myths. Glob Environ Change. 11(4):261–269.

Li A, Wu J, Huang J. 2012. Distinguishing between human-induced and climate-driven vegetation changes: a critical application of RESTREND in inner Mongolia. Landscape Ecol. 27(7):969–982.

Newell-Price JP, Whittingham MJ, Chambers BJ, Peel S. 2013. Visual soil evaluation in relation to measured soil physical properties in a survey of grassland soil compaction in England and Wales. Soil Tillage Res. 127:65–73.

Petorelli N, Vij KO, Mysterud A, Gaillard JM, Tucker CJ, Stenseth NC. 2005. Using the satellite-derived NDVI to assess ecological responses to environmental change. Trends Ecol Evol. 20(9):503–510.

Pimentel D, Harvey C, Resosudarmo P, Sinclair K, Kurz D, Pimentel E, Turner BL, Geist HJ, Agbola SB, Angelsen A, Bruce JW, Coomes OT, Dirzo R, Fischer G, Folke C, et al. 2001. The causes of land-use and land-cover change: moving beyond the myths. Glob Environ Change. 11(4):261–269.

Petorelli N, Vij KO, Mysterud A, Gaillard JM, Tucker CJ, Stenseth NC. 2005. Using the satellite-derived NDVI to assess ecological responses to environmental change. Trends Ecol Evol. 20(9):503–510.

Pimentel D, Harvey C, Resosudarmo P, Sinclair K, Kurz D, McNair M, Crist S, Shpritz L, Fitton L, Saffouri R, et al. 1995. The causes of land-use and land-cover change: moving beyond the myths. Glob Environ Change. 11(4):261–269.

Pettorelli N, Vij KO, Mysterud A, Gaillard JM, Tucker CJ, Stenseth NC. 2005. Using the satellite-derived NDVI to assess ecological responses to environmental change. Trends Ecol Evol. 20(9):503–510.

Peplau C, Don A. 2013. Sensitivity of soil organic carbon stocks and fractions to different land-use changes across Europe. Geoderma. 192:189–201.

Post WM, Kwon KC. 2000. Soil carbon sequestration and land-use change: processes and potential. Global Change Biol. 6(3):317–327.

Reed BC, Brown JF, VanderZee D, Loveland TR, Merchant JW, Ohlen DO. 1994. Measuring phenological variability from satellite imagery. J Veg Sci. 5(5):703–714.

Soussana J-F, Soussana J-F, Loiseau P, Vuichard N, Ceschia E, Balesdent J, Chevallier T, Arrouays D. 2004. Carbon cycling and microbial community composition across land use and incubation temperature. Soil Biol Biochem. 39(3):744–756.

Stenseth NC. 2005. Using the satellite-derived NDVI to assess ecological responses to environmental change. Trends Ecol Evol. 20(9):503–510.

Thiim AK. 2003. The causes and spatial pattern of land degradation risk in southern Mauritania using multitemporal development goals (SDGs). [accessed 2019 December 9]. https://www.iass-potsdam.de/sites/default/files/files/land_and_soil_indicators_proposal.pdf
AVHRR-NDVI imagery and field data. Land Degrad Dev. 14(1):133–142.

United Nations. 2012. The future we want. Resolution A/RES/66/288 adopted by the General Assembly. [accessed 2019 December 9]. https://sustainabledevelopment.un.org/content/documents/733FutureWeWant.pdf.

United Nations. 2015. Transforming our world: the 2030 agenda for sustainable development. Resolution A/RES/66/288 adopted by the General Assembly.

United Nations Convention to Combat Desertification. 2013. Refinement of the set of impact indicators on strategic objectives 1, 2 and 3. Recommendations of the ad hoc advisory group of technical experts. ICCD/COP(11)/CST/2. [accessed 2019 December 9]. https://www.unccd.int/sites/default/files/sessions/documents/ICCD_COP11_CST_2/cst2eng.pdf.

United Nations Convention to Combat Desertification. 2016. Land degradation neutrality target setting – a technical guide. [accessed 2019 December 9]. https://knowledge.unccd.int/sites/default/files/2018-08/LDN%20TS%20Technical%20Guide_Draft_English.pdf.

United States Environmental Protection Agency. 1986. Method 9080 – cation exchange capacity of soil (ammonium acetate). [accessed 2019 December 9]. https://www.epa.gov/sites/production/files/2015-12/documents/9080.pdf.

Yengoh GT, Dent D, Olsson L, Tengberg AE, Tucker CJ. 2014. The use of the normalized difference vegetation index (NDVI) to assess land degradation at multiple scales: a review of the current status, future trends, and practical considerations. Washington: Scientific and Technical Advisory Panel of the Global Environment Facility (STAP/GEF).

Zimmermann M, Meir P, Silman MR, Fedders A, Gibbon A, Malhi Y, Urrego DH, Bush MB, Feeley KJ, Garcia KC, et al. 2010. No differences in soil carbon stocks across the tree line in the Peruvian Andes. Ecosystems. 13(1):62–74.