Soil erodibility estimation for modeling soil erosion in hillslope and watershed scales

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1. INTRODUCTION

Soil erosion is one of the most important environmental problems in both regional and global scales (Jianping, 1999). Prevention of soil erosion is of paramount importance in the management and conservation of natural resources such as soil and water (Sadeghi et al., 2007). For conservation of soil, predicting soil erosion and proper evaluation of the main erosional factors in an area of interest is important for developing the management practices (Cai et al., 2020).

For prediction of soil erosion, soil erodibility is an important parameter (Rodríguez et al., 2006). Soil erodibility is defined as resistance of detachment by rain splash and transport by overland flow (Morgan, 1995). Erodibility factors is used for original universal soil loss equation (USLE) for estimating annual soil erosion (Wischmeier and Smith, 1978). For RUSLE, Jiang et al. (2020) used 0.026 Mg h MJ^{-1} mm^{-1} for shrub (*Sophora davidii*) and 0.057 Mg h MJ^{-1} mm^{-1} for coniferous (*Platycladus orientalis*) of K factors. Soil erodibility of the coniferous forest soil in the Upper Yangtze River basin, China was estimated 0.03 Mg h MJ^{-1} mm^{-1} (Wang and Li, 2013). Other finding using Soil and Water Assessment Tool (SWAT) soil erodibility was estimated for watershed covered by evergreen forest located in Western India ranged from 0.1 to 0.4 Mg h MJ^{-1} mm^{-1} (Ganasri and Ramesh, 2016).

Soil properties and soil erosion is spatially vary depending on complex interactions between geology, topography, vegetation within watersheds (Jiang et al., 2020). For instance, Liang et al. (2017) showed soil porosity ranged from 0.38 to 0.84 at the natural forest because of different land units such as gully area, side slope and valley-head slope. The spatial variation of soil organic carbon in 38.5 ha forested catchment ranged from 1.4 to 44% because of spatial variation of soils in the catchment consist mainly of Cambisols and Planosols in the higher and drier regions and Gleysols and Histosols in the riparian zone located around the stream (Gottselig et al., 2017). Based on previous findings of spatial variability, soil erodibility factor (K-factor) for model prediction can also likely spatially vary.

Most of the previous studies consider the soil erodibility parameter as a single parameter within a given area. Cai et al. (2020) used multiple K value 0.009 Mg h MJ^{-1} mm^{-1} for *Pinus armandii*, 0.012 Mg h MJ^{-1} mm^{-1} *Arborvitaes* (Thuja spp.) and 0.002 Mg h MJ^{-1} mm^{-1} for Sea-buckthorn. For examining the uncertainty of the K factor, Wang et al. (2001) conducted analysis of the spatial prediction and uncertainty with the sample data using sequential Gaussian simulation. Hence, soil erodibility mapping is important to the assessment of soil erosion control and the map has great relevance for many activities related to land use planning and guidance for the soil conservation practices and modeling (Bonilla and Johnson, 2012).

Evaluating spatial variability of soil condition in forested area is essential for improving the predictability of soil erosion. Thus, the objective of this study is (1) investigating the spatial variability of soil properties including from valley bottom to ridge line in headwater catchment; and (2) estimating spatial variability for soil erodibility factor for model parameter of soil erosion.

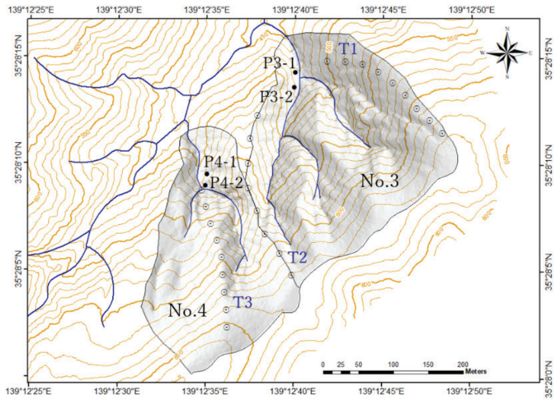

Figure 1. Study area location map and transect soil sampling

2. STUDY AREA AND METHODOLOGY

The study was conducted in two forested headwater catchments No.3 (7ha) and No.4 (5ha) of the Oborasawa Watershed, located in Tanzawa Mountains in the eastern part of the Kanagawa Prefecture (Fig.1). The mean temperature and annual precipitation are 12^0 C and 3000 mm, respectively (Oda et al., 2013), and 41-66% of all precipitation falls from June to October. The dominant forest vegetation of the upper catchment is 20 to 30 years coniferous trees *(Cryptomeria japonica* and *Chamaecyparis obtusa*). Lower part is covered by deciduous broadleaf trees (*Cercidiphyllum japonicum* and *Aesculus turbinate*) (Oda et al., 2013).

For examining spatial variability of soil condition, we selected three transects from the valley bottom to the ridge in October 2020. Sampling locations were selected based on equally spaced plots through the catchment with about 30 m interval. We collected the soil sample using 30 cm liner sampler (DIK 110C; Daiki Rika Kogyo Co., Ltd.). We also collected two soil samples for the analysis of soil organic carbon and particle size analysis (size distribution of particles in a given soil sample). The slope gradient was computed using digital elevation model DEM and the vegetation ground cover was measured using 50 cm x 50 cm quadrant. Vegetation ground cover was classified as little litter coverage with no vegetation (class1), intermediate litter coverage with no vegetation (class2), large litter coverage with no vegetation (class3), little vegetation coverage <40% (class4), intermediate vegetation coverage 40-80% (class5) and large vegetation coverage >80% (class6) (Braun-Blanquet, 1964). The vegetation was collected from each quadrant and the dry mass weight was measured using 105^0c oven at 24hrs.

30 cm liner soil samples were separated into each 5 cm and analyzed soil bulk density, soil texture and organic matter contents. Soil bulk density was estimated based on (Blake and Hartge, 1986). Soil particle size is classed as >2 mm, 1-2 mm, and <1 mm sieve. For fine particles, percentages of soil particle sizes in samples were determined by weight. For fine particles (≤1 mm), we used a laser particle size analyzer (SALD-2300). The soil texture was described in terms of the percentages of sand (0.05-2 mm), silt (0.002-0.05 mm) and clay (<0.002 mm) (Osman, 2013). The soil organic carbon was calculated as the mass difference between oven-dry and ashes weights (Zhang and Wang, 2014).

Soil erodibility factor K was estimated based on the following equation proposed by Sharpley and Williams (1990).

$$K = \left(0.2 + 0.3 * exp\left[-0.0256 * m_s * \left(1 - \frac{m_{silt}}{100}\right)\right]\right) * \left(\frac{m_{silt}}{m_c + m_{silt}}\right)^{0.3}$$

$$* \left(1 - \frac{0.25 * orgC}{orgC + exp[3.72 - 2.95 * orgC]}\right)$$

$$* \left(1 - \frac{0.7 * \left(1 - \frac{m_s}{100}\right)}{\left(1 - \frac{m_s}{100}\right) + exp\left[-5.51 + 22.9 * \left(1 - \frac{m_s}{100}\right)\right]}\right) * 0.1317$$

where: m_s, m_{silt}, m_c and $_{org}c$ are sand, silt, clay and organic carbon content respectively (%).

3. RESULTS

The altitude of the sampling plots was ranged varied from 473 to 653 m a.s.l. and the gradient of the sampling plots was ranged 7 to 61^0. The bulk density was ranged from 0.2 to 1.5 g/cm^3 (mean: 0.7; SD: 0.2) and soil organic carbon content (SOC) was ranged from 3.0 to 20.0% (mean: 9.1; SD: 3.2). Sand content ranged from 74.9 to 94.3% (mean: 88.1; SD: 4.0), while silt content was from 5.6 to 25.2% (mean: 11.9; SD: 3.9). Clay content was low with ranging from 0 to 1.8% (mean: 0.1; SD: 0.2).

For ridgeline, mean soil bulk density was 0.5 g/cm^3 ranged from 0.2 to 0.6 g/cm^3. Mean value of SOC in ridgeline was 10.7% from 5.8 to 19.7% of ranges. For valley side (near stream channels), mean soil bulk density was 0.8 g/cm^3 ranging from 0.3 to 1.5 g/cm^3, while SOC ranged from 3.1 to 20.0% with 8.5% mean values (SD: 3.0). Sand content ridgeline was 79.5 to 93.4% with mean values of 87.6%, while mean sand contents of soil near streams was 88.4 (74.9 to 97.6). Mean silt contents in ridge line was 12.3%, while mean silt contents in valley side was 11.5%.

The vegetation ground cover was varied from 30 to 85.1% (mean: 61.3; SD: 16.6), the litter dry mass was varied from 0 to 1,760.8 g/m^2 (mean: 340.3; SD: 446.7) and the plant dry biomass was varied from 0.02 to 116.5 g/m^2 (mean: 26.1; SD: 42.6). Where high ground cover was found in catchment No.3 of three sampling plot of the ridgeline and in one sampling plot of the valley side, whereas in the catchment No.4 high ground cover was found in one sampling plot of the ridgeline and in one sampling plot of the valley side.

Estimated soil erodibility was ranged from 0.01 to 0.02 Mg h MJ^{-1} mm^{-1}. The mean soil erodibility near the ridgeline was 0.01 Mg h MJ^{-1} mm^{-1} with ranging from 0.01 to 0.02 Mg h MJ^{-1} mm^{-1} and the mean in valley side was 0.01 Mg h MJ^{-1} mm^{-1} ranging from 0.01 to 0.02 Mg h MJ^{-1} mm^{-1}.

4. DISCUSSIONS

We found that our estimated soil erodibility (K) was rather smaller estimation by Lin et al. (2019) ranged from 0.002 to 0.01 Mg h MJ^{-1} mm^{-1} in coniferous and broadleaf forests of 39.1 km^2 areas. Small K value by Lin et al. (2019) was found in the upstream of the watershed. Cai et al. (2020) found that soil erodibility value in Loess Plateau forested area (0.1ha) in north western region of China ranged from 0.009 to 0.012 Mg h MJ^{-1} mm^{-1}.

Our findings showed that soil erodibility factors did not vary within locations, although vegetation and litter cover is different. Because of less variation of the soil morphology and soil types, K factors were also did not varied although measured soil erosion in plot scale differed. Thus, for modeling soil erosion using RUSLE, single K-value can be also effective within in catchments. Hence, for accurate measurement of soil erosion, vegetation and litter cover is important because these factors physical proceed soil by splash and overland flow (Wang et al., 2020). Therefore, we will develop complete model for including vegetation and litter covers for prediction of soil erosion.

5. CONCLUSIONS

The soil erodibility model was estimated for each sampling plots across the hillslope and the catchments. The result based on this approach did not vary within locations by some uncertainty of the soil erodibility estimation. Therefore, efficient supplementary inputs are necessary for further improving the precision of K value predictions such as vegetation and litter cover which are important factors for the soil erosion.

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Keywords: *Soil erodibility, Soil properties, Organic carbon, Soil erosion, Watershed scale*