**Nutrient and Isotopic Dynamics of Litter Decomposition from Different Land Uses in Naturally Restoring Taihang Mountain, North China**

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**Abstract:** Litter decomposition is a prominent pathway for nutrient availability and management in terrestrial ecosystems. An in-situ litter decomposition experiment was carried out for different land use types along an elevation gradient in the Taihang Mountain area restored after heavy forest degradation in the past. Four land use types, i.e., cropland, shrubland, grassland, and forest, selected randomly from a 300–700 m elevation were investigated for the experiment using the litter bag technique. Litter mass loss ranged from 26.9% (forest) to 44.3% (cropland) varying significantly among land use types. The initial litter quality, mainly N and C/N, had a significant effect on the litter loss rate. The interaction of elevation $\times$ land use types $\times$ time was significant ($p < 0.001$). Litter nutrient mobility ($K > P \approx N > C$) of the decomposing litter was sporadic with substantial stoichiometric effects of $C/N$, $N/P$, and $C/P$. The residual litters were enriched in $^{15}N$ and depleted in $^{13}C$ as compared to the initial litter. Increment of N, P, and $\delta^{15}N$ values in residual litter indicates that, even in the highly weathered substrate, plant litter plays a crucial role in conserving nutrients. This study is a strong baseline for monitoring the functioning of the Taihang Mountain ecosystem restored after the complete destruction in the early 1990s.

**Keywords:** litter decomposition; nutrient dynamics; nitrogen and carbon isotopes; different land use types; restoration

1. **Introduction**

Litter decomposition serves as a key pathway for the return of organic matter and nutrients to the soil from above-ground plant residues and hence maintains soil fertility in terrestrial ecosystems. Nutrient dynamics in different decomposition phases have a strong influence on (a) the nutrient sustainability of the terrestrial ecosystems [1,2], (b) organic matter accretion in the soil [3,4], and (c) nutrient budgets [5,6]. Supply of the major nutrients needed for forest growth is influenced by varying rates of litter decomposition followed by mineralization [7–9], providing the majority of the energy required to regulate biogeochemical processes governed by saprophytic organisms [6]. The rate of litter decomposition is determined by both biotic (soil macro-fauna and microbial diversity) and abiotic (litter chemistry, soil properties, and microclimate) factors [3]. Litter decomposition is predominantly vital for forests that are extremely reliant on natural fertilization, growing on highly...
weathered soils/rock systems, where nutrient supply is constrained, and phosphorus and nitrogen cycling pace are slow but stable [10]. Land use conversion of forest results in the disruption of ecological processes including nutrient losses [11]. As a result, forest growth dynamics and shifts in species composition regulate the decomposition pattern during successions stages. Decomposition in young forests is determined by nutrient content while that of older forests is determined by microbial characteristics [12]. All the mechanisms that determine the availability of nutrients in aggrading forests are based on the recycling of nutrients through litter decomposition [1,4,13]. A complete understanding of litter decomposition and the extent of nutrient mobility is crucial for the development of an effective plan in order to sustain nutrient resources, especially in secondary forests that regenerate after decades [10,14]. Litter decomposition has been broadly studied in cool temperate forests [1,5], whereas information about secondary forests and other vegetation [12,15] in their recovery stage, developed on highly rugged parent material, is sparse. A better understanding of the temporal pattern of litter nutrients is essential for adopting the best management strategies for maintenance, conservation, and the growth of restoring (secondary) forests [10].

The natural abundance of carbon (\(\delta^{13}C\)) and nitrogen (\(\delta^{15}N\)) isotopes and their dynamism during different phases of decomposition have been useful in understanding and indicating the progression of litter decomposition, nutrient release/accumulation, and accessibility [13,16]. Variation in the isotopic composition of residual litter and its relationship with the rate of litter loss and litter nutrient dynamics are considered the best inherent tracers for understanding each step in the decomposition process. The isotopic signature is lowered due to the depletion of \(^{13}C\) in recalcitrant lignin and its discriminating preservation-associated loss of labile \(^{13}C\)-enriched compounds (sugar, starch, and cellulose). In addition, microbial processes enrich carbon with \(^{13}C\) in relation to bulk litter [17]. \(^{15}N\)-depleted labile N loss (uptake, leaching, and nitrification), together with \(^{15}N\)-enriched microbial biomass can be traced. Since bacteria have great potential for immobilization of nitrate depleted in \(^{15}N\), its dominance can decrease \(\delta^{15}N\) signatures [18,19]. These findings have provided an important insight into several decomposition studies in the past to track the alteration of the isotopic signatures of C and N during in- and out-flow of the immobilization-mineralization phases to examine the decomposition process [6,20,21].

Primary and old-growth forests in the hilly area of the Taihang Mountains in North China have disappeared significantly in the past decades due to various anthropogenic activities such as deforestation, cutting, tilling, logging, and grazing. In 1999, the ‘Grain for Green’ (GfG) program was initiated by the Chinese government to recover vegetation through compensation of farmers in terms of free grains and cash so as to encourage the conversion of cultivated land with slopes greater than 25° to forest or pasture land [22,23]. In order to better understand the uniqueness of impaired forest ecosystems and to document the information in support of vegetation recovery in the lower Taihang Mountain areas, the Hilly Ecological Experimental Station (HEES) was established by the Chinese Academy of Sciences. The forest is still in the recovery stage with existing crop fields in the foothill region. Various studies regarding the change in soil chemical composition have been conducted [22,24]. However, the nutrient dynamics in different phases of litter decomposition have not yet been documented for this part of the study area. The variation in vegetation composition has been examined [25], but the role of the change in retaining the soil nutrient content of the aggrading forest ecosystem has not yet been documented. It is imperative to understand changes in the nutrient pattern of different litter types along with the shift in their isotopic properties to know the current situation of an aggrading forest and track its functioning further.

Therefore, it is important to examine litter decomposition and to analyze the nutrient dynamics of different land use systems that developed after disappearance of the primary forest due to intensive deforestation and grazing activities in the Taihang Mountain area, growing under inhibited resource conditions. The following hypotheses were tested: (1) the leaf litter decomposition of different land use types differs significantly even in a narrow elevation gradient; (2) nutrient dynamics in decomposing litter could indicate whether the restored ecosystem is in a nutrient conservative stage; (3) isotopic variation among litter types and during decomposition stages is an important indicator of nutrient
status. If proved, these hypothesis could support our main objective to examine the nutrient dynamics and changes in the natural abundance of carbon ($\delta^{13}$C) and nitrogen ($\delta^{15}$N) isotopes of various land use systems along an elevation gradient through an in-situ litter decomposition experiment in the Taihang Mountains.

2. Materials and Methods

2.1. The Study Area and Sampling

The study was carried out at the Hilly Ecosystem Experimental Station of the Taihang Mountains (114°15′50″ E, 37°52′44″ N), Chinese Academy of Sciences. The experimental station is located in the Niujiazhuang Catchment (an area of 9.3 km$^2$) situated 50 km southwest of the capital city, Shijiazhuang, Hebei Province, China (Figure 1). The altitude ranges from 247 to 1040 m. The average annual temperature is 13 °C, with a minimum temperature of −1.6 °C and a maximum temperature of 29.1 °C. The average annual precipitation is 560 mm with a mean annual evaporation of 1934 mm (Figure 2). The annual precipitation of the catchment is 390–750 mm, and 67.8% of rain is received from June to September [22].

The parent rock of the study area is composed mainly of granite, granite porphyry, limestone, sandstone, and shale. The mountainous cinnamon soil, formed mainly by calcareous soil from the
weathering of gneissic granitoids and partly by limestone and shale, is the predominant soil type. The soil profile, with a depth of 120 cm mostly on north-facing slopes and 20 cm on south-facing slopes, is poorly stratified with a thin O horizon (0–2 cm) and poorly developed A (3–10 cm) and B horizons (20–70 cm) with abundant rock chips (highly weathered). The soil is barren, with a weak potential for retaining fertilizer and water, and is highly permeable [22].

The study was conducted on the north-facing slope of the Taihang Mountains extended to the highest altitude of 700 m with a slope >25°. Land use types at each elevation were identified through stratified random sampling starting from the cropland at the foothill, followed by shrubland, grassland, and forest. The study sites were located at every 100 m on an elevation gradient from 300 to 700 m a.s.l. Altogether, four land use types—cropland, shrubland, grassland and forest—along five elevations were considered based on the dominant vegetation (Tables 1 and S4).

Table 1. Site characteristics at five sites along elevation gradient.

<table>
<thead>
<tr>
<th>Site No.</th>
<th>Elevation</th>
<th>Land Use Types</th>
<th>Latitude</th>
<th>Longitude</th>
<th>Dominant Vegetation</th>
<th>Management Practices</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>700 m</td>
<td>Shrubland</td>
<td>N37°53′56.3″</td>
<td>E114°15′50.3″</td>
<td>Vitex negundo</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Grassland</td>
<td>N37°53′56.8″</td>
<td>E114°15′50.5″</td>
<td>Themeda japonica</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Forest</td>
<td>N37°53′57.8″</td>
<td>E114°15′49.6″</td>
<td>Pinus tabulaeformis</td>
<td>-</td>
</tr>
<tr>
<td>2</td>
<td>600 m</td>
<td>Forest</td>
<td>N37°54′02.6″</td>
<td>E114°15′46.4″</td>
<td>Pinus tabulaeformis</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Grassland</td>
<td>N37°54′07.0″</td>
<td>E114°15′40.9″</td>
<td>Themeda japonica</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Shrubland</td>
<td>N37°54′07.5″</td>
<td>E114°15′38.6″</td>
<td>Vitex negundo</td>
<td>-</td>
</tr>
<tr>
<td>3</td>
<td>500 m</td>
<td>Grassland</td>
<td>N37°54′09.4″</td>
<td>E114°15′32.1″</td>
<td>Themeda japonica</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Shrubland</td>
<td>N37°54′09.4″</td>
<td>E114°15′32.1″</td>
<td>Vitex negundo</td>
<td>-</td>
</tr>
<tr>
<td>4</td>
<td>400 m</td>
<td>Shrubland</td>
<td>N37°54′24.0″</td>
<td>E114°15′26.3″</td>
<td>Vitex negundo</td>
<td>- Irrigation and fertilizer application</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Cropland (Millet)</td>
<td>N37°54′25.2″</td>
<td>E114°15′26.7″</td>
<td>Millet</td>
<td>- Irrigation and fertilizer application</td>
</tr>
<tr>
<td>5</td>
<td>300 m</td>
<td>Shrubland</td>
<td>N37°57′14.3″</td>
<td>E114°15′09.1″</td>
<td>Punica granatum</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Cropland (Millet)</td>
<td>N37°54′12.4″</td>
<td>E114°15′07.2″</td>
<td>Millet</td>
<td>- Irrigation and fertilizer application</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Cropland (Maize)</td>
<td>N37°54′13.8″</td>
<td>E114°15′07.6″</td>
<td>Maize</td>
<td>- Irrigation and fertilizer application</td>
</tr>
</tbody>
</table>

2.2. Initial Litter Collection, Decomposition, and Soil Sampling

Freshly fallen leaf and <2 cm diameter twigs were collected randomly from different land use systems in October 2016, air dried, and stored indoors until the beginning of the in-situ decomposition experiment. Different plots were delineated in each ecosystem, with elevation starting from 300 to 700 m at an interval of 100 m. Nylon litterbags were used for studying litter decomposition [5]. Mixed air-dried (8 g oven-dry mass at 80 °C) litter samples of the respective land use types were placed in 10 × 15 cm litterbags prepared from 1 mm mesh nylon cloth. We placed an adequate number of litterbags (4 × 13 sites × 6 times = 312 litter bags) in all 13 sites representing four land use types distributed at five elevations. Litter bags were placed in the respective sites for in-situ decomposition in February 2017, tied to a nylon rope. Both ends of the rope were tied to the branches of trees, or pegs were used to prevent the loss of bags, their dispersion due to wind and heavy snow accumulation in winter. On each collection date (bi-monthly), starting from April 2017 to February 2018, litterbags (at least three per site to compensate the missing bags) were collected. The litterbags were carefully and separately packed in Ziploc bags to avoid the loss of fine particles and carried to the lab. The adhering soil particles and extraneous materials were brushed free; then, specimens were gently washed under running deionized water, after which they were air dried for 2 weeks and then oven dried at 80 °C until a constant mass was recorded. The oven dried mass of each litterbag was weighed and rounded to the nearest ±0.01 g. The litter loss percent was measured with reference to the initial oven dry weight.

Soil samples (0–10 cm) were collected from the respective land use types at the time of litter bag collection, i.e., every two months. The collected samples were carefully transported to lab to analyze pH and moisture content.
2.3. Chemical and Isotopic Analyses

The litter samples were ground to a fine powder. Total nitrogen (N) and available phosphorus (P) were analyzed by sulfuric acid and hydrogen peroxide digestion [26], followed by titration to determine the N content. The ammonium molybdate ascorbic acid method was used to determine the P content of the digest. The concentration of K was measured using an atomic absorption spectrometer.

The soil pH was measured using a pH meter (Mettler Toledo LE438) in soil to distilled water suspension of ratio 1:2.5 and a moisture content by drying fresh soil in oven at 105 °C for 24 h.

\[
\text{Soil moisture (\%)} = \frac{\text{Initial weight of soil (g)} - \text{Oven dry weight of soil (g)}}{\text{Oven dry weight of soil (g)}} \times 100
\]  

(1)

The litter carbon (C) and nitrogen (N) concentrations were also analyzed with an Elemental Analyzer: Variopyro cube (Elementar, Germany). The natural abundance of δ13C and δ15N was measured using Stable Isotope Ratio Mass Spectrometry (Isoprime100, Elementar, Germany). The carbon and nitrogen isotope ratios are expressed in delta notation (δ) as

\[
\delta (\text{‰}) = \left( \frac{R_{\text{sample}}}{R_{\text{standard}}} - 1 \right) \times 1000
\]  

(2)

where \(R_{\text{sample}}\) and \(R_{\text{standard}}\) refer to either \(^{13}\text{C}:/^{12}\text{C}\) or \(^{15}\text{N}:/^{14}\text{N}\) ratios of sample and standard material, respectively. The stable isotope ratio values are expressed in parts per mil (‰) relative to international standards: Vienna Pee Dee Belemnite (VPDB) for carbon isotopes and atmospheric nitrogen for nitrogen isotopes. Analytical precision for carbon isotopes is ±0.2‰ and ±0.4‰ for nitrogen isotopes.

2.4. Data Calculation and Analysis

2.4.1. Litter Mass Loss and Decomposition Rate

Percentage of mass loss was calculated by

\[
\text{Mass loss (\%)} = \frac{\text{Initial mass \(X_0\)} - \text{Final mass \(X_t\)}}{\text{Initial mass \(X_0\)}} \times 100
\]  

(3)

The dry litter mass percentage remaining at time \(t\) was calculated according to Bragazza et al. [18]:

\[
\text{Remaining mass (\%)} = 100 - \left[ \frac{(X_0 - X_t)}{X_0} \right] \times 100
\]  

(4)

The decay rate constants (\(k\)) for the different litter types were calculated using a negative exponential model:

\[
k = -\ln(X_t/X_0)/t
\]  

(5)

where \(X_0\) is the initial dry weight of the litter, \(X_t\) is the weight of the residual litter after time \(t\), and \(t\) is the sampling duration (in years). Time required for 50\% (\(t_{50}\)) and 95\% (\(t_{95}\)) weight loss were calculated according to Olson [27]:

\[
t_{50} = 0.693/k
\]

\[
t_{95} = 5/k
\]

2.4.2. Statistical Analyses

Significant differences of all the parameters for initial litter were analyzed using one-way ANOVA. Multiple comparison tests were carried out using Fisher’s least square difference (LSD) analysis (post hoc) for differences in litter loss percent, decomposition rate, nutrient dynamics, and litter chemistry across different ecosystems and elevation. The effects of elevation, land use types, and time (duration) on litter decomposition and other nutrient parameters were analyzed using a mixed model (two-way
and three-way ANOVAs). Changes in C, N, C/N, $\delta^{13}C$, and $\delta^{15}N$ with time were analyzed using repeated measure ANOVA. The relationship among litter nutrients as well as between litter mass loss and nutrient change throughout the decomposition process were tested by linear regression. The relationships between litter mass losses, chemical content, and isotopic composition were tested using Pearson’s correlation analysis. The climatic parameters such as temperature and rainfall were used for correlation and regression analyses. All the tests were considered significant at $p < 0.05$ level. Analyses were done in IBM SPSS statistics 23 and graphs were drawn using MS Excel.

The nutrient mobility of decomposing litter types were ranked based on maximum variation between initial and final concentrations either in the form of increase or decrease.

3. Results

3.1. Decomposition and Initial Litter Chemistry

A year-long in-situ litter decomposition experiment demonstrated a phasic litter decomposition pattern. Initially, litter decomposition was slow until August, with mass loss <10% for all land uses. The decomposition process accelerated after six months. The result here indicated a rapid increase in litter mass loss during the post rainy season, i.e., August–October, and remained nearly stable with the onset of a cold dry winter (accompanied by snow fall) (Figure 3). The litter mass after one year declined (on average) to 40, 38, 37, and 29% of the original mass for cropland, shrubland, grassland, and forest, respectively. Cropland at 400 m showed the highest mass loss (42.93% ± 0.75), and the forest at 700 m showed the lowest percent decrease in litter mass (26.92% ± 0.86). The litter mass loss for different land use types was ranked as cropland > shrubland ≥ grassland > forest. The results showed significant ($F = 20.15, p < 0.001$) variation among land use types. For each land use type, the litter mass loss and decomposition rate varied significantly along the elevation gradient ($F = 8.17, p < 0.001$); however, there was no consistent pattern in relation to the increase in elevation. The decomposition period and elevation gradient had significant effects on the decomposition rate of all land use types ($p < 0.001$) except for the forest ($p = 0.06$). In addition, the results showed a significant interaction of land use types, elevation, and time period ($p < 0.01$) with litter mass loss.

The annual decay constant (k/year) varied significantly among the elevations and ranged from 0.56/year (Millet400) to 0.31/year (Fr700). The time required for 50% and 95% litter decay also varied significantly except for cropland (Table 2). The time required for 50% litter decay ranged from 1.2 years (Millet400) to 2.2 years (Forest), and that for 95% decay ranged from 8.92 years (Millet400) to 15.99 years (Fr700). The decay rates for different land uses were as follows: cropland ≥ shrubland ≥ grassland > forest.

The initial litter nutrient (N, P, K, and C) concentration showed significant variation among the four land use types. The initial nitrogen (N) concentration ranged from 0.75–1.95%. Litter N differed significantly among the land use types ($F = 10.85, p < 0.001$) and along the elevation gradient ($p < 0.01$). There was an increase in the concentration of litter N with elevation, except for grassland and forest (Table S1). The phosphorus (P) concentration (ranged from 0.01–0.097%) did not vary significantly ($F = 1.64, p =0.2$) among land uses. However, the difference was significant with elevation ($p = 0.001–0.003$). The concentration of potassium (K) in the litter of different land use types, except for the forest, differed significantly ($F = 340, p < 0.001$) along their respective elevation gradients. The K content ranged from 0.23% (Fr700) to 1.9% (Millet400) and increased with elevation. The concentration of carbon (C) varied significantly for each land use type with the elevation gradient. The forest had the highest C concentration (Fr700 = 49.56%), while the concentration was lowest in cropland (Millet300 = 39.51%). Similarly, the C/N ratio of all land use types showed significant differences with elevation ($p <0.001–0.02$). Initial $\delta^{13}C$ and $\delta^{15}N$ values varied significantly among the land use types and elevations ($p < 0.001$). The result showed that litter at the cropland site was highly enriched in $^{15}N$ as compared to other land uses. The $\delta^{13}C$ value was higher in the cropland and grassland as compared to shrubland and forest (Table S1).
Figure 3. Cumulative litter mass loss percentage of different land use types during one year decomposition period: (a) cropland, (b) shrubland, (c) grassland, and (d) forest; n = 3.

Table 2. Annual decomposition constant (k) of mass loss and time required to achieve 50% ($t_{50}$) and 95% ($t_{95}$) decomposition (in years), n = 3.

<table>
<thead>
<tr>
<th>Land Use Types</th>
<th>ID</th>
<th>k</th>
<th>$t_{50}$</th>
<th>$t_{95}$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cropland</td>
<td>Maize300</td>
<td>0.49 (0.009)</td>
<td>1.40 (0.03)</td>
<td>10.14 (0.19)</td>
</tr>
<tr>
<td></td>
<td>Millet300</td>
<td>0.49 (0.042)</td>
<td>1.44 (0.11)</td>
<td>10.40 (0.83)</td>
</tr>
<tr>
<td></td>
<td>Millet400</td>
<td>0.56 (0.013)</td>
<td>1.24 (0.03)</td>
<td>8.92 (0.21)</td>
</tr>
<tr>
<td></td>
<td>F</td>
<td>2.77</td>
<td>2.43</td>
<td>2.43</td>
</tr>
<tr>
<td></td>
<td>p</td>
<td>0.14</td>
<td>0.17</td>
<td>0.17</td>
</tr>
<tr>
<td>Shrubland</td>
<td>Sh300</td>
<td>0.44 (0.012) (^a)</td>
<td>1.56 (0.04) (^a)</td>
<td>11.30 (0.30) (^a)</td>
</tr>
<tr>
<td></td>
<td>Sh400</td>
<td>0.55 (0.020) (^b)</td>
<td>1.25 (0.04) (^b)</td>
<td>9.01 (0.32) (^b)</td>
</tr>
<tr>
<td></td>
<td>Sh500</td>
<td>0.47 (0.015) (^c)</td>
<td>1.48 (0.05) (^ac)</td>
<td>10.73 (0.36) (^ac)</td>
</tr>
<tr>
<td></td>
<td>Sh600</td>
<td>0.51 (0.008) (^bc)</td>
<td>1.37 (0.02) (^bc)</td>
<td>9.89 (0.15) (^bc)</td>
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<tr>
<td></td>
<td>Sh700</td>
<td>0.46 (0.009) (^ac)</td>
<td>1.51 (0.03) (^c)</td>
<td>10.90 (0.22) (^c)</td>
</tr>
<tr>
<td></td>
<td>F</td>
<td>11.35</td>
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<td>10.72</td>
</tr>
<tr>
<td></td>
<td>p</td>
<td>&lt;0.01</td>
<td>0.001</td>
<td>0.001</td>
</tr>
<tr>
<td>Grassland</td>
<td>Gr500</td>
<td>0.49 (0.014) (^a)</td>
<td>1.42 (0.04) (^a)</td>
<td>10.26 (0.31) (^a)</td>
</tr>
<tr>
<td></td>
<td>Gr600</td>
<td>0.42 (0.007) (^b)</td>
<td>1.63 (0.02) (^b)</td>
<td>11.79 (0.18) (^b)</td>
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<tr>
<td></td>
<td>Gr700</td>
<td>0.49 (0.009) (^a)</td>
<td>1.41 (0.03) (^a)</td>
<td>10.16 (0.20) (^a)</td>
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<tr>
<td></td>
<td>F</td>
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<td>14.58</td>
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<td>p</td>
<td>0.007</td>
<td>0.005</td>
<td>0.005</td>
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<tr>
<td>Forest</td>
<td>Fr600</td>
<td>0.38 (0.007) (^a)</td>
<td>1.84 (0.04) (^a)</td>
<td>13.29 (0.26) (^a)</td>
</tr>
<tr>
<td></td>
<td>Fr700</td>
<td>0.31 (0.012) (^b)</td>
<td>2.22 (0.08) (^b)</td>
<td>15.99 (0.61) (^b)</td>
</tr>
<tr>
<td></td>
<td>F</td>
<td>20.39</td>
<td>16.43</td>
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<tr>
<td></td>
<td>p</td>
<td>0.01</td>
<td>0.015</td>
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</tr>
</tbody>
</table>

Values are the mean (± SE) of three replicates for annual decay rate (k), time required for 50% decay ($t_{50}$), and 95% decay ($t_{95}$). Different superscripts (a, b, c) indicate significant differences between initial and final litter chemistry ($p < 0.05$). $F$ represents one way ANOVA result, and $p$ is the level of significance. “Sh”, “Gr”, and “Fr” stand for shrubland, grassland, and forest with the respective elevation gradients alongside.
There was a positive correlation between litter mass loss and initial N, N/P, and C/P values of the land use types (Table 3). However, the correlation was mostly negative with initial P, K, C, and C/N. The forest showed a strong correlation of litter mass loss with the initial litter nutrient concentration. The concentration of P, K, and C expressed an inverse relationship with the decomposition pattern. The stoichiometry of initial litter C, N, and P was correlated with the litter mass loss; there was a negative correlation with C/N and positive correlations with N/P and C/P ratios. Cropland litter with low C/N but high N/P and C/P decomposed faster as compared to other litter types.

### Table 3. Pearson’s correlation coefficients of litter mass loss (%) and initial litter chemistry.

<table>
<thead>
<tr>
<th>Land Use Types</th>
<th>Initial N</th>
<th>Initial P</th>
<th>Initial K</th>
<th>Initial C</th>
<th>Initial C/N</th>
<th>Initial N/P</th>
<th>Initial C/P</th>
<th>δ¹⁵N</th>
<th>δ¹³C</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cropland</td>
<td>0.55</td>
<td>-0.47</td>
<td>0.19</td>
<td>-0.88 **</td>
<td>-0.33</td>
<td>0.68 *</td>
<td>0.65</td>
<td>-0.09</td>
<td>0.48</td>
</tr>
<tr>
<td>Shrubland</td>
<td>0.39</td>
<td>-0.47</td>
<td>-0.15</td>
<td>-0.06</td>
<td>-0.48</td>
<td>0.26</td>
<td>0.26</td>
<td>0.73 **</td>
<td>-0.59 *</td>
</tr>
<tr>
<td>Grassland</td>
<td>0.52</td>
<td>-0.87 *</td>
<td>-0.03</td>
<td>-0.89 **</td>
<td>-0.73 *</td>
<td>0.65</td>
<td>0.68 *</td>
<td>-0.91 **</td>
<td>-0.32</td>
</tr>
<tr>
<td>Forest</td>
<td>0.94 **</td>
<td>-0.88 **</td>
<td>-0.16</td>
<td>-0.91 **</td>
<td>-0.95 **</td>
<td>0.91 *</td>
<td>0.88 *</td>
<td>0.92 **</td>
<td>-0.92 **</td>
</tr>
</tbody>
</table>

**Correlation is significant at the 0.01 level. * Correlation significant at the 0.05 level.

#### 3.2. Nutrient Dynamics during Decomposition

Litter mass loss was positively correlated with litter N, P, and soil moisture during decomposition. The correlation was mostly negative for K, C, C/N, N/P, C/P, soil pH, and the isotopic composition (δ¹⁵N) of decomposing litter of different land use types (Table S3). The nitrogen content of the decomposing litter changed significantly over time for cropland ($r^2 = 0.41–0.79$, $p = <0.001–0.002$), shrubland ($r^2 = 0.41–0.42$, $p = 0.002–0.004$), and grassland ($r^2 = 0.25–0.053$, $p = <0.001–0.02$). By the end of the decomposition process, N concentration of Millet300 and Millet400 was reduced by 12.9% and 7.8%, respectively, of the initial concentration. However, for Maize300, N concentration was observed to increase by 67.5%. The N concentration of forest and shrubland litter increased by 46.8% and 33.5%, respectively, of the initial concentration (Table 4). Shrubland at 400 and 500 m showed a gradual accumulation of nutrients initially followed by a sudden release in October and accumulation in the later phase (Figure S1b). The concentration of N was decreased in Sh600 over 10 months but increased in the later phase. Sh300 and Sh700 did not show any changes in concentration during the process, but the residual litter differed significantly from the initial litter N concentration (Table 4). Nitrogen was released initially and then accumulated in the later phase of litter decomposition of the forest. In the case of grassland, at a 500 m elevation, the N concentration was reduced by 26.1% compared with the initial litter, but increases of 18.1% and 10.2% of the initial concentration were observed at 600 and 700 m elevations, respectively. N was released by grass litters until August, followed by an accumulation during October and then a return to the lag phase (Figure S1).

The phosphorus concentration of the decomposing litter changed significantly over time for land use types at different elevations, such as grassland ($r^2 = 0.2–0.46$, $p = 0.001–0.04$), shrubland at 600 m ($r^2 = 0.45$, $p = 0.001$), and forest at 600 m ($r^2 = 0.32$, $p = 0.008$). By the end of the year-long decomposition experiment, the litter P was increased by >50% for all the land uses as compared to their initial concentration (Table 4). There was a sharp fluctuation in the temporal pattern with an alternation of increases and decreases in P concentrations (Figure S2). Cropland showed a sudden increase in phosphorus initially, followed by a gradual loss, and this was significant at $p < 0.01$. Shrubland showed an accumulation of phosphorus in the initial phase of litter decomposition followed by a release in the later phase in a significant manner ($p < 0.02$). The concentration increased gradually in the initial phase, then decreased drastically, and increased again in the later phase of decomposition for grassland. Similarly, the forest showed an initial accumulation of P followed by a release in the later phase of decomposition ($p < 0.05$) and then an accumulation at the end. A linear regression of litter mass loss against changing P concentration showed a significant relationship for the forest ($p = 0.01$).
The potassium (K) concentration decreased gradually and consistently throughout the decomposition period for all land use types (Figure S3). The change in concentration during the process was consistent as compared to N and P, where the fluctuation was abrupt. The K concentration in decomposing litter decreased by 69.5, 55.9, 14.9, and 6.04% in cropland, grassland, shrubland, and forest (Table 4). The temporal change was significant for shrubland ($r^2 = 0.29–0.51, p = 0.01$ to $<0.01$). Grassland across all three elevation gradients showed a gradual release of K. The difference was weakly significant ($r^2 = 0.19–0.22, p = 0.03–0.04$). A linear regression of K against litter mass loss was significant for cropland ($r^2 = 0.34, p < 0.001$), shrubland ($p = 0.001$), forest ($r^2 = 0.22, p = 0.004$), and grassland ($r^2 = 0.55, p < 0.001$). The nutrient mobility was ranked as cropland > grassland > shrubland > forest.

There was a decrease in carbon (C) concentration of the residual litter from all land use types (Table 4). The concentration decreased by 27.7% (cropland), 5.56% (shrubland), 6.1% (grassland), and 6.04% (forest). The regression analysis did not fit significantly for the change in the C concentration of cropland except Millet400 ($p < 0.01$) and shrubland except Sh700 ($p < 0.01$). However, the concentration varied significantly in grassland and forest ($p$ range: 0.003–0.03). The change in C concentration was slow and gradual in the initial phase followed by a considerable release in the later phase (Figure S4). The relationship between litter mass loss and C dynamics during different phases of decomposition was consistent as compared to N and P, where the fluctuation was abrupt. The K concentration in decomposing litter decreased by 69.5, 55.9, 14.9, and 6.04% in cropland, grassland, shrubland, and forest (Table 4). The relationship between litter mass loss and C dynamics during different phases of decomposition was significant for cropland ($p < 0.001$) and shrubland ($p = 0.01$) while insignificant for the rest. The dynamics of the C/N ratio of decomposing litter were linearly significant with the litter mass loss for forest ($p = 0.01$) and cropland ($p = 0.07$). Similarly, the relationship of both the N/P and C/P ratios of the decomposing litter with litter mass loss was significant for all land use types at $p < 0.01$. By the end of the decomposition experiment, the change in concentration in relation to initial litter differed significantly for most land use types (Table 4).

<table>
<thead>
<tr>
<th>Land Use Types</th>
<th>ID</th>
<th>Remaining Mass (%)</th>
<th>N (%)</th>
<th>P (%)</th>
<th>K (%)</th>
<th>C (%)</th>
<th>C/N</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cropland</td>
<td>Maize300</td>
<td>61.05 (0.58)</td>
<td>0.75 (0.01)</td>
<td>0.08 (0.004)</td>
<td>1.46 (0.01)</td>
<td>39.58 (0.02)</td>
<td>53.08 (0.34)</td>
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<tr>
<td></td>
<td>Millet300</td>
<td>63.40 (0.72)</td>
<td>1.26 (0.02)</td>
<td>0.05 (0.01)</td>
<td>1.88 (0.03)</td>
<td>39.51 (0.01)</td>
<td>31.38 (0.47)</td>
</tr>
<tr>
<td></td>
<td>Millet400</td>
<td>57.07 (0.75)</td>
<td>1.10 (0.01)</td>
<td>0.15 (0.005)</td>
<td>0.67 (0.02)</td>
<td>25.16 (0.05)</td>
<td>22.95 (0.35)</td>
</tr>
<tr>
<td>Shrubland</td>
<td>Sh300</td>
<td>64.21 (0.75)</td>
<td>1.70 (0.02)</td>
<td>0.02 (0.002)</td>
<td>1.90 (0.003)</td>
<td>42.54 (0.03)</td>
<td>26.02 (0.34)</td>
</tr>
<tr>
<td></td>
<td>Sh400</td>
<td>57.34 (1.15)</td>
<td>1.57 (0.03)</td>
<td>0.21 (0.01)</td>
<td>0.47 (0.02)</td>
<td>31.23 (0.06)</td>
<td>19.93 (0.42)</td>
</tr>
<tr>
<td>Grassland</td>
<td>Gr500</td>
<td>61.38 (0.9)</td>
<td>0.75 (0.05)</td>
<td>0.07 (0.007)</td>
<td>0.47 (0.006)</td>
<td>44.9 (0.04)</td>
<td>43.90 (2.12)</td>
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<tr>
<td></td>
<td>Gr600</td>
<td>65.42 (0.43)</td>
<td>1.3 (0.03)</td>
<td>0.11 (0.007)</td>
<td>0.34 (0.04)</td>
<td>36.89 (0.02)</td>
<td>28.05 (0.54)</td>
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<tr>
<td></td>
<td>Gr700</td>
<td>61.42 (0.6)</td>
<td>1.16 (0.02)</td>
<td>0.01 (0.003)</td>
<td>0.27 (0.02)</td>
<td>41.87 (0.03)</td>
<td>36.22 (0.61)</td>
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<tr>
<td>Forest</td>
<td>Fr600</td>
<td>68.63 (0.5)</td>
<td>1.95 (0.0)</td>
<td>0.10 (0.002)</td>
<td>0.26 (0.008)</td>
<td>44.65 (0.006)</td>
<td>22.9 (0.003)</td>
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<tr>
<td></td>
<td>Fr700</td>
<td>73.08 (0.86)</td>
<td>2.15 (0.01)</td>
<td>0.14 (0.01)</td>
<td>0.3 (0.02)</td>
<td>43.9 (0.03)</td>
<td>20.16 (0.05)</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Land Use Types</th>
<th>ID</th>
<th>Remaining Mass (%)</th>
<th>N (%)</th>
<th>P (%)</th>
<th>K (%)</th>
<th>C (%)</th>
<th>C/N</th>
</tr>
</thead>
</table>

Values are the mean ($±$SE) of three replicates for remaining mass, initial litter chemistry, and final litter chemistry. Different superscripts indicate significant differences between initial and final litter nutrients ($p < 0.05$). "Sh", "Gr", and "Fr" refer to shrubland, grassland, and forest with the respective elevation gradients alongside.
3.3. Isotopic Changes during Decomposition

The in-situ litter decomposition experiment demonstrated increases in the N-isotopic ($\delta^{15}N$) composition despite rapid fluctuations of enrichment and depletion observed throughout the process. There was a strong negative correlation between $\delta^{15}N$ and litter loss rate for all the land use types, i.e., the $\delta^{15}N$ value was higher when decomposition was slow and dropped sharply as the litter mass loss accelerated (Table S3). Decomposing litter became highly enriched in $^{15}N$ during summer (June, 2017) and showed a gradual decline thereafter. A sudden increase in the $\delta^{15}N$ value coincided with the elevated temperature (Figure 4), but the correlation was weakly positive. The linear regression between litter mass loss and change in $\delta^{15}N$ was significant ($p < 0.001$). The land use types did not show significant relationships between $\delta^{15}N$ and climatic parameters. Similarly, the relationship of the $\delta^{15}N$ value with N concentration was linearly significant for Millet400 ($r^2 = 0.5$, $p < 0.01$), Sh700 ($r^2 = 0.77$, $p < 0.01$) and forest ($r^2 = 0.36$, $p < 0.01$) (Figures S6a and S8a). Pearson’s correlation between $\delta^{15}N$ and N was highly positive for Millet400 ($r = 0.7$) and Sh700 ($r = 0.88$) but highly negative for Fr600 ($r = -0.59$) and Fr700 ($r = -0.88$). The phosphorus content showed a strong positive correlation with $\delta^{15}N$ during the decomposition period. Similarly, the linear regression analysis of $\delta^{15}N$ with the C/N ratio resulted in a significant relationship for the forest ($r^2 = 0.53$, $p < 0.01$), but the remaining land use types showed insignificant results (Figure S6b).

![Figure 4](image_url). Temporal pattern of litter $^{15}N$ and mean monthly temperature for different land use types: (a) cropland, (b) shrubland, (c) grassland, and (d) forest; $n = 3$.

The $\delta^{13}C$ value of the residual litter was slightly lower with reference to the initial litter, though the change was not significant (Figure S5). The relationship of litter mass loss and $\delta^{13}C$ dynamics was significant only for the forest (Table S3). Regression analysis (linear fit) between climatic factors such as temperature and rainfall did not show significant results with respect to the litter loss rate and $\delta^{13}C$. The correlation between the C-isotope and N was significant for Sh600 ($r = -0.89$), Gr600 ($r = -0.45$), and Fr600 ($r = -0.45$) but non-significant for the rest. The relationship of $\delta^{13}C$ with litter C and the C/N ratio was not significant (Figure S7b,c). Similarly, there was a significant relationship between
 δ¹³C and δ¹⁵N for Gr600 ($r^2 = 0.55$), Millet400 ($r^2 = 0.32$), Fr600 ($r^2 = 0.28$), and Sh300 ($r^2 = 0.29$), $p < 0.01$ (Figure S8c).

4. Discussion

4.1. Litter Decomposition in Different Land Use Types along the Elevation Gradients

In this experiment, we observed three different phases of litter mass loss: (1) an initial phase (Feb–Aug) characterized by a slow decline, (2) an intermediate phase (Aug–Oct) characterized by a rapid decline in litter mass coincident with the post rainy season, and (3) a later phase (Oct–Feb), followed by slow and consistent mass loss, mainly in the winter (dry) season. Slow decomposition in the initial phase and last phase is attributed to extremely cold weather as snow accumulates in our study area, inhibiting the microbial population and their enzymatic activities. In a previous study, rapid litter loss occurred in the initial phase [10], mostly in summer followed by the rainy season. However, our result showed no rapid change immediately with the onset of summer and the rainy season; the cumulative effect of temperature and precipitation during the months of August–October might be a factor responsible for the drastic increase in mass loss (from approximately 15–45%). Moreover, the area received a significant amount of rainfall in August and October 2017 (Figure 2), which might have favored the litter decay process. The rapid loss might be due to the interactive effects of (a) climatic conditions that promoted the population and activity of soil fauna [5,10], (b) the development of microbial and fungal communities [21,28], and (c) rapid labile C loss through the C pool [5,13,29]. A higher range of decomposition rates was found in the wet season than in the dry season because of site microclimatic factors, i.e., rainfall and temperature, and the substrate condition [30]. This is because decomposition is particularly seasonal in nature. The freezing and thawing events in winter might change the litter quality, contributing to the decay process as the majority of the labile material is decomposed [31–33].

Cropland (Millet400) had the lowest mass disappearance time, and the forest showed the longest time required for 50% ($t_{50}$) and 95% ($t_{95}$) mass loss. Gautam et al. [10] recently reported similar results for the mass disappearance rate ($t_{50}$ and $t_{95}$) of different litter types in order of pinus twigs > grass > pinus leaves > shrub in cool temperate forest. However, the value for each litter type in our study was considerably higher as compared to the results of previous work, which can be attributed to the mixed litter (leaves + twigs) used in this study. Mixed litter has different chemical and physical properties, altering the resource quality and physical habitat complexity that result in changes in decomposition rates along with the abundance of decomposer organism and their activities. When the litter is mixed, mass loss often increases due to the transfer of nutrients between the decomposing species, mediated by the fungal mycelia [34–37]. Mass loss from twig litter was lower, as compared to leaf litter for pine litter [10] and beach litter [20], and also in four tree species of sub-alpine coniferous forests [7]. The difference in the decomposition rate was observed to be higher in the cropland; the reason is attributed to the management practices and alteration of conditions during cultivation, including irrigation and fertilizer input (Table S4). The decomposition of plant litter mainly is hierarchically controlled by factors such as, from most significant to the least, climate, litter quality, and soil organisms [38]. The fate and process of litter decomposition are strongly dependent upon the initial chemistry and nutrient content of the litter. In various previous studies, the N concentration and C/N ratio of the initial litter have been highlighted as the most reliable predictors of litter decomposition [3,32,34,39]. Good litter quality with a high initial N concentration [40] and low C/N ratio [41] generally yields a fast decomposition rate. In this study, Sh700, Sh600, and Millet400 had higher N concentrations and lower C/N ratios in the initial litter compared to other land use types that were characterized by comparatively faster decomposition throughout the study period. Similarly, litter with the lowest N concentration and highest C/N (Maize300 and Fr700) was found to decompose slowly. The litter loss for cropland and grassland was not correlated significantly with
The annual decay constants (k/year) of litter types from different ecosystems followed the pattern reported in previous studies: shrub > grass > pine [10]; the values are congruent with k values reported for cool temperate species [42], but are lower than those observed in temperate broad-leaved [1] and coniferous forests [43]. The result indicates that the decomposition rate varies, even between plant species growing under similar lithological and climatic conditions. Gautam et al. and Osono et al. [10,42] noticed differences in the decay constant of functionally dissimilar vegetation growing adjacent to each other, which is similar to our results. These kinds of variations in the decay rate among different litter types could be due to the effect of small-scale topographical variation in moisture, humus, and edaphic distinctiveness [44]. The pine forests in our study site were in a well-drained ridge (with reduced soil moisture) as compared to other land use types (shrubland, grassland, and cropland). Similarly, a high decomposition rate and litter loss rate were consistent with flat, poorly drained soils, shallow water tables, and elevated moisture in the studied cropland.

Elevation gradients can be considered natural and long-term analogues for climate change. The loss of litter mass as mediated by the microclimatic factors has been corroborated by natural gradient studies [34,38,45,46]. This finding is supported by the results of this study, where each site showed significant variation along the elevation gradient despite having a similar substrate condition and vegetation type. Temperature and moisture in combination are the most important climatic controls affecting litter decomposition rates, and there is an increase in litter mass loss in a warming environment if the soil moisture is sufficiently high [34,45,46]. The mass loss and rate of decomposition varied among the studied land uses, across and within the elevation gradient, which might be due to the litter quality, the presence of varying amounts of water-soluble compounds, physiochemical properties of the litter, and the presence of thick waxy cuticles (pine) [47,48]. A study by Xu et al. [49] showed a significant decrease of mass loss with increased elevation, which is comparable with our result. The lowest rate in the forest was attributed to the surface area of pine needles and the cuticles inhibiting the supply of metabolites to the microbial community, which was expected to be the accelerating factor for cropland, shrubland, and grassland. In the later phase, the influence of the labile N content and C/N ratio declined over time. The predominance of recalcitrant metabolites that are resistant to enzymatic (microbial) and environmental degradation was suspected to reduce litter mass loss in the later phase [5,10,42]. Litter decomposition can be principally divided into the early phase (before 20–40% mass loss), determined by the initial litter chemistry, while the later phase is mainly controlled by the decomposition of the lignin and the microbial community [34,50,51]; our study has demonstrated the early phase of decomposition.

4.2. Nutrient Dynamics during Decomposition

We observed significant (p < 0.05) changes in N, P, K, C, and C/N ratio during decomposition. Decomposition in the land use types was noticed to differ significantly (p < 0.001) along their respective elevation gradients throughout the decomposition period. However, the change did not follow a consistent pattern of an incline or decline due to rapid fluctuation in the nutrient release and accumulation of decomposing litter from the substrate. Diverse microbial communities colonizing the different litter types along with alteration in mineralization and immobilization might be the cause for change observed during the entire process [10]. Litter decomposition was positively correlated with N and P dynamics of the decomposing litter, indicating an increase in the concentration with decomposition rate. The increase in the N concentration of the residual litter was as follows: cropland > shrubland > forest > grassland, where the highest gain by cropland might be attributed to fertilizer application. However, the convergence of N in other natural land use types is probably due to initial litter chemistry and microbial interaction. The correlation between the changing concentration of N and P was positive, suggesting synergetic interaction between them during litter decomposition. Similarly, the P concentration increased sharply in the initial decomposition phase with a sudden decline in
August followed by a sharp increase in the later phase. The order of increased P concentration was cropland > shrubland > grassland > forest. The increased concentration of N and P in different phases is attributed to immobilization by bacteria and/or the transfer of nutrients to the decomposing litter from humus or mineral soil via fungal hyphae [5,52]. The microbial activity enhanced by favorable temperature and precipitation might be a cause for the sharp changes seen during the summer–autumn (post rainy) season. The N mobilization occurs in low-quality litter (N content < 20 mg g$^{-1}$ or C/N > 25), whereas high-quality litter (N content > 20 mg g$^{-1}$ and C/N < 25) releases N during decomposition as suggested by Myers et al. [53]. Similar results were observed in this study as cropland (except Maize300) and Gr500 released N, while there was convergence of N in other systems. The decrease in the C/N ratio during the decomposition period (but not for grassland) indicated N retention and respiratory C removal within the decomposer microbial population and the litter [54]. The increase in N and P during the decomposition period was attributable to initial high lignin and low N and P concentrations [2,52]. Nutrient accretion might also be due to formation of lignin-like recalcitrant substances with N and P [5,8], based on the lignin/N and/or lignin/P ratios of the decomposing litter. Retention and loss of phosphorus during litter decay could be predicted from the initial N/P ratio, i.e., there is a loss of P if the N/P ratio of initial litter is 10–20 and a gain if the initial ratio >20. A change in the N/P ratio during litter decomposition has recently been acknowledged as a determinant of nutrient release and limitation in ecosystems [55], which is supported by our findings as litter gained P if the N/P ratio was >20 for all the land use types.

In this study, the carbon concentration was increased initially but released gradually in the later phase. Loss of C from different land uses was ranked as cropland > grassland > forest > shrubland. A strong positive correlation between changing C and N concentrations of the decomposing litter in this study suggests that C mass loss was promoted by increasing N concentration. Besides decomposition dynamics, C/N, N/P, and C/P ratios are the factors controlling C loss [49]. A change in the potassium concentration was consistent throughout the decomposition period. A sharp decline in concentration was observed in the initial period of decomposition. An initial rapid drop off in the concentration can be credited to the nature of K (highly mobile), making it susceptible to leaching due to its loose ionic bonding with structural tissues [5], and slow release in the later phase can be attributed to refractory components [56]. This kind of dynamics in the K concentration is comparable with previous findings that reported potassium as the most mobile cation during litter decomposition [8,57,58]. However, in our study, cropland exhibited the highest initial K content and a consistent decline thereafter, and a cropping pattern (wheat and maize) and different management practices (irrigation, fertilizer implication) during decomposition were ascribed as the probable causes. The accelerated concentration in cropland is credited to fertilizer application, and the rapid decline is due to activities such as soil management, irrigation, and other disturbances, which enhance the release of the cation from residual litter. Mineralization and immobilization of decomposing litter are greatly influenced by the alteration in the microbial community [59] due to seasonal variation in the temperature and moisture content [5] of the intact surface. The nutrient loss occurred remarkably after the onset of summer followed by the rainy season as the result of a favorable environment enhancing the growth of the microbial community and its activities. The soil of the region, being highly weathered and fragile, are frequently eroded in the rainy season, which is supposed to be another cause for nutrient loss through surface runoff or leaching.

4.3. Dynamics of the Natural Abundance of $\delta^{15}$N and $\delta^{13}$C during Decomposition

We noticed a gradual change in $\delta^{15}$N values across all the ecosystems as decomposition proceeded. Litters from cropland were slightly enriched while other land use types were $^{15}$N depleted. The $\delta^{15}$N value for different litter types showed a declining mode during the first four months of decomposition followed by a sudden increase during the month of June and then a consistent decrease in the later phase. The sharp increase in $\delta^{15}$N value in the month of June was positively correlated with temperature and negatively with rainfall. This indicates that an increase in temperature
supports $^{15}$N enrichment of decomposing litter. Previous studies reported a decrease in $\delta^{15}$N during decomposition [10,52], while some observed $^{15}$N enrichment [13,16] as well. However, our result showed a strong negative correlation between litter loss and $\delta^{15}$N, suggesting that litter is depleted in $^{15}$N as the decomposition reaction accelerates and is enriched as the loss rate slows down. The litter sampled in study sites was enriched in $^{15}$N during the later phase (after one year) of the decomposition cycle. The $\delta^{15}$N increase in the residual litter was explained as the outcome of microbially induced fractionations or the preferred retention of refractory compounds enriched with $\delta^{15}$N. However, preferred retention is uncertain due to little variation in the component litter fraction from whole tissue N [16,60]. This indicates that fractionation of isotopes due to microbial activity might be a debatable source of $\delta^{15}$N variation. Microbial dominance during decomposition also plays an important role in the release and retention of $\delta^{15}$N; a fungally dominated decay process results in $^{15}$N enrichment [21], whereas a bacterially dominated decomposition process depletes $\delta^{15}$N signatures [18]. The decreased value of $\delta^{15}$N, even while the C/N ratio was relatively high, might be linked with microbial requirement being fulfilled by the decomposing litter according to [61]. Another study suggested that higher C availability enhances fractionation during N assimilation, causing a decline in $^{15}$N enrichment [17]. Linear regression of litter $\delta^{15}$N value with N was not significant for most of the land uses, but the relationship was significant with C/N. This relationship signifies that $\delta^{15}$N discrimination in our study is due to preferential retention of $^{15}$N-enriched substrates or the additive effect of the discriminatory process on the $\delta^{15}$N content. The linear regression implies isotopic discrimination from directional loss from the substrate [16]. A long-term litter decomposition study is required to better identify the isotopic trend and associated mechanisms for better understanding. Microbial analysis is essential to support the convergent nutrient cycling and the divergent isotopic pattern of litter decomposition [49].

We noticed a negligible shift in $\delta^{13}$C values of different land uses during decomposition. However, the $\delta^{13}$C value of residual litter was slightly lower than that of the initial litter. This depletion of $^{13}$C in residual litter could be described by the preferential degradation of more labile and isotopically heavier constituents with a consequent accretion, in residual litter, of $\delta^{13}$C-depleted carbon compounds [62]. In most cases, the $\delta^{13}$C value declined slightly during the first two-month period followed by a slight increase and then remained unchanged thereafter (except for the forest). Isotopic changes are caused by the microbial mixing of C with residual litter while undergoing transformation [16]. Studies on $\delta^{13}$C isotopic changes during C flow in different phases of decomposition via fractionation have not shown a consistent pattern. Wedin et al. [63] and Xu et al. [13] reported enrichment in residual litter, Fernandez et al. [64] observed depletion, and a negligible shift was reported by Ngao and Cotrufo [65]. A study by Connin et al. [16] suggested that the early litter decay process characterizes the isotopic discrimination based on the observation of two isotopic shifts for the same region with two different species—one shift leading to enrichment and the other resulting in depletion. The slight changes (enrichment) observed in our study are attributed to substrate quality and selective discrimination by the microbial community against $\delta^{13}$C [64]. Similar to our result, an increase in $\delta^{13}$C value was observed at the end of summer, indicating an increase in residual litter with rising air temperature [66]. This is further supported by the findings of previous research that showed decreased $^{13}$C abundance of respired CO$_2$ and consequent $^{13}$C enrichment of residual litter through increased temperature [66,67]. However, in the forest ecosystem, $^{13}$C was depleted during the decomposition process; the result can be attributed to the transfer of C fractions enriched in $\delta^{13}$C from the residual litter to the underlying soil surface through microbial interaction or leaching. A continuous quality theory was proposed by Ågren et al. [68]: the temporal trend of $\delta^{13}$C in decomposing substrates is controlled by the partitioning of $^{13}$C and $^{12}$C among chemical fractions with varying quality and properties of the decomposers (assimilation efficiency and growth rate). A decrease in $\delta^{13}$C value occurs because of the isotope effect due to litter quality, but an increase occurs due to decomposers. The combined effect of these factors could lead to a variable $\delta^{13}$C discrimination pattern as a function of decomposition. The study even
suggested that litter quality is the most important determinant of $\delta^{13}$C discrimination [68]. The lack of initial lignin data in our study limits us from analyzing the relationship.

5. Conclusions

During a year-long litter decomposition experiment, the decomposition rate started slowly in the initial stage followed by a rapid intermediate phase and continued with a consistent, slow trend. The annual decay constant and mass loss rates demonstrated site-specific response, following the sequence of cropland $>$ shrubland $\geq$ grassland $>$ forest. Rapid litter loss coincided with initial litter quality (high N and low C/N ratio) and climatic factors. Nutrient mobility during the decay process showed an initial release phase (loss by transfer/leaching) followed by a slight increase (absorption from the soil surface). The litter stoichiometry (C/N, C/P, and N/P) showed a significant effect on litter decomposition and nutrient mobility. The nutrient dynamics of the decomposing litter were in the order of K $>$ P $\approx$ N $>$ C, in terms of the highest percentage change in concentration of residual litter with reference to initiation litter concentration. N and P were accumulated by the decomposing litter while the rest of the nutrients was released. Increases in N and P concentrations in the residual litter showed the prevalence of nutrients in conservation mode controlling the loss (erosion, leaching) from the highly fragile mountain ecosystem. The overall trend of litter loss and nutrient dynamics is linked with biotic and abiotic factors. The abrupt changes throughout the decomposition cycle did not result in any specific conclusion regarding net accumulation or release of nutrients. Therefore, a long-term decomposition experiment and microbial analyses are essential to draw conclusions about the functioning of forest ecosystems in the Taihang Mountains. Slight changes in $\delta^{13}$C and an increase in $\delta^{15}$N values in residual litter indicate the convergent nature of isotopic discrimination. The non-discernable change (irregular pattern) in the natural abundance of $\delta^{13}$C in all the land use types suggests a negligible exchange of C between the residual litter and the underlying microbial community or a mutual balance between the $^{13}$C-enriched substrate (feedback mechanism) interfering with distinct fractionation. The relationship between litter loss with $\delta^{15}$N change explains the microbially mediated fractionation or the preferential retention of $\delta^{15}$N from the enriched substrate.

Overall, our results show that the mountain ecosystem is leaning towards a conservative nutrient mechanism through a slow rate of decomposition and gradual nutrient recycling. Accumulation of N, P, and $\delta^{15}$N indicates that, even under the highly weathered substrate condition, different vegetation types are functioning well to conserve nutrients. Variations among litter from similar land use systems with elevation in our study provide a clear picture of the microclimatic preference of the decomposition mechanism coupled with microbial activity. However, a long-term experiment and microbial analysis are required to draw conclusions regarding the performance of aggrading forest ecosystems such as that of the Taihang Mountains in terms of nutrient recycling.

Supplementary Materials: The following are available online at http://www.mdpi.com/2071-1050/11/6/1752/s1, Figure S1: Nitrogen dynamics of the decomposing litter, Figure S2: Phosphorus dynamics of the decomposing litter, Figure S3: Potassium dynamics of the decomposing litter, Figure S4: Carbon dynamics of decomposing litter, Figure S5: $\delta^{13}$C changes of the decomposing litter in one year, Figure S6: Relationship of litter N-isotope with; (a) Nitrogen, (b) C/N ratio, and (c) N/P ratio, Figure S7: Relationship between litter C-isotope with; (a) N-isotope, (b) Carbon and (c) C/N ratio, Figure S8: Relationship between; (a) Litter nitrogen and N-isotope, (b) Litter carbon and C-isotope and (c) N-isotope and C-isotope, Table S1: Initial litter chemistry of different land use types along elevation, Table S2: Three way ANOVA: Land use types*Elevation*Time, Table S3: Pearson’s correlation coefficients (r values) between litter mass loss and nutrient dynamics decomposing litters from different land use types, Table S4: Fertilizer application and management practices in cropland.

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