EFFECTS OF DEGRADATION ON PHOSPHORUS CYCLING IN ALPINE WETLAND OF TIBETAN PLATEAU

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Abstract. Phosphorus (P) cycle plays a vital role in maintaining the structure and function of wetland ecosystems. However, research knowledge on high-altitude wetland ecosystems is very poor. In the present study, plant communities, soil P fractions and other properties and their transformation were investigated under the influence of four types of degradation categories, namely non-, light-, medium-, and severe degraded (ND, LD, MD, and SD, respectively) wetlands of Tibetan Plateau. The degradation of wetlands causes succession in the plant communities from wet to dry mesophytic communities, significantly reducing plant coverage and biomasses. As a result, soil water content (SWC), soil organic carbon (SOC), and soil total nitrogen (TN) contents decrease with increasing degradation. Plant P accumulation usually showed a decreasing trend with increasing levels of degradation. Soil total P, available P, inorganic P, Labile inorganic P, P mineralization rate, and alkaline phosphatase (ALP) concentration increased from non-degradation to light degradation and then declined until severe degradation. The redundancy analysis (RDA) ranking further revealed that labile organic P is the main source of soil available P and soil ALP play important role in P transformation. These results suggest that there prevails a nonlinear change on soil P pool and turnover rate to wetland degradation degree. The study thus provides a mechanistic understanding of alpine wetland degradation driven soil P dynamics and cycling, which would facilitate the restoration of nutrient-based degraded wetlands.
Keywords: *wetland degradation, plant community, plant P accumulation, soil P fraction, P mineration*

Introduction

Wetlands are among the world’s most productive and valuable ecosystems (Ding et al., 2021). Despite covering only 1.5% of the Earth’s surface, wetlands provide 40% of the ecosystem services worldwide (de Vicente, 2021). Biogeochemical cycles are the key driver of wetland ecosystem services. The phosphorus (P) cycle is an important component of nutrient cycling in wetland ecosystems and has a key regulatory role in the structure, processes, and function of the ecosystems. Wetland P cycling is susceptible to the effects of utilization patterns as well as climate change (Childers and Noe, 2007). With the rapid growth of human populations, global wetlands have been suffering from serious degradation or loss due to pollution, reclamation, urbanization, and land use pattern changing, and so forth (Zhang et al., 2019). Wetland degradation has a potential influence on the P cycling that govern important wetland ecosystem services (de Vicente et al., 2021; Zhang et al., 2019). Therefore, it is an urgent task to understand the changes in P cycling during wetland degradation processes, which can contribute effectively to wetland management and restoration.

Wetland degradation is often accompanied by a decline in aquifer and soil water content (Li et al., 2022). A change in the aquifer’s hydrological conditions can induce a significant change in plant composition and biomass (Zhang et al., 2019). This affects accordingly the amount of P accumulation in above- and belowground plant biomass

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(Zhang et al., 2019; Menon and Holland, 2014). The plant P pool in wetlands also depends upon the decomposition and mineralization rates of dead plant material which help adding organic P into the soil system (Menon and Holland, 2014). These processes are subjected to plant and soil water condition (Li et al., 2022; Zak et al., 2014). Due to an oxidizing environment after wetland degradation, microbe-induced inorganic P solubilization, and organic P mineralization become active and, as a result, P transformation may be enhanced (de Vicente, 2021). Low groundwater levels promote the decomposition of organic matter and loss of humus and peat layers in soil, and may simultaneously promote the conversion of soil P to aerobic processes resulting in soil P available reduction (Robinson and Niedermeier, 2009). Therefore, the response of P biogeochemical cycling may dependent on wetland type and degradation degree. Although degradation may alter P biogeochemical processes in wetland ecosystems, little is known about the impact of degradation on P storages in plant and soil P transformation in alpine wetland.

The Tibetan Plateau is well known because world's largest alpine wetland ecosystems are present and widely distributed in this area (Yang et al., 2022). They are fragile ecosystems and characterized by their high-altitude locations and low temperatures. Over the past few decades, due to the influence of human activities (drainage and overgrazing) and climate change, the alpine wetlands have suffered quite a severe degradation, which has resulted not only a decline in plant production but also a deterioration of the ecological environment (Yang et al., 2022). Many studies have examined the responses of soil nutrients, physicochemical properties, and plant communities of degraded alpine wetlands (Yang et al., 2022; Ren et al., 2013; Gao et al., 2018). However, a comparison of P stocks in ecosystem components, soil P fraction and transformation at different stages of degradation has not yet been conducted. Therefore, the aims of the present study were (i) to measure the storage of P in plant and soil along different wetland degradation gradients; and (ii) to clarify the effect of alpine wetland degradation on soil P fraction and transformation. This study will improve our understanding of the degradation impacts on alpine wetland, and provide data to support the theoretical basis for scientifically evaluating the P budget of degraded alpine wetland ecosystems.

Materials and methods

Study site

The study sites are located at the Yueliangwan wetland park (32°78′ N, 102°48′ E, 3500 m a.s.l.) in Hongyuan County in Sichuan Province, China. This area has a continental plateau monsoon climate. The mean annual temperature is 1.1°C. The highest monthly mean temperature is 10.9°C in July and the lowest is –10.3°C in January. The mean annual precipitation is 753 mm, with approximately 86% received from May until September. The major vegetation types are meadow and marsh-meadow, dominated by *Carex muliensis*, *Kobresia setchuanensis*, *Elymus nutans* and *Poa pratensis*. The soil types mainly include peat-, swamp-, and meadow soil (Gao et al., 2018).

Experimental design and sampling

The study area was divided into four degraded sites with different degrees: the non-degradation (ND), light degradation (LD), moderate degradation (MD), and severe degradation (SD), according to the vegetation cover, biomass, and species composition

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(Yang et al., 2022). The study sites are distanced by 100 m from each other having similar elevations and peat soil parent material. It is assumed that the site conditions of the undegraded sites, reflect the conditions of degraded sites before degradation. The basic information of each degradation gradient is shown in *Table 1*.

At each site, three 10 × 10 m^2 plots, including three diagonal sampling quadrats (1 × 1 m^2), with over 10 m apart from each other were established in early August 2021. In each quadrat, plant species and coverage were identified and recorded. Aboveground biomass was clipped to ground level as living and dead biomass. Dead biomass samples included standing dead biomass and litter. Root biomass was measured by collecting 3 soil cores (20 cm dia) from depths of 0-20 cm in each plot, which were co-located with the aboveground biomass measurement quadrates. These cores were immediately washed over a 1-mm mesh screen to remove soil. All plant samples were oven-dried for 48 h at 70°C and weighed afterwards.

Five soil cores from each plot were extracted (5 cm dia, and 20 cm in depth) and thoroughly mixed, and then each sample was divided into two parts. One part of the sample was air dried for measuring pH, SOC, TN, and P fraction. The other part was kept at field moisture content at 4°C for soil enzyme activities.

Sample analysis

The soil organic carbon (SOC) content was determined by the $K_2Cr_2O_7$ method (Lu, 1999). Soil and plant N was measured by the micro-Kjeldahl method using a Kjeltec 8200 Autoanalyser (Foss Electric A/S, Denmark), respectively (Lu, 1999). Soil and plant P was measured by spectrophotometer (Agilent 8453 UV–Vis, Agilent Technologies, Inc.) after wet digestion with H_2SO_4 and $HClO_4$ (Yakutina, 2011). Available soil P was determined using the Olsen method by extracting samples with $NaHCO_3$, and determining P colorimetrically using molybdate (Turner et al., 2012). Soil inorganic P was extracted by H_2SO_4, followed by colorimetric measurement of phosphate ions in the filtrates (Bowman and Cole, 1978). The soil organic P (OP) was defined as the difference between the total and inorganic P (IP) (Ivanoff et al., 1998). Soil water content was measured by the oven-drying method (105°C for 48 h). Operationally defined labile Pi and labile Po forms were extracted with $NaHCO_3$ (Grunwald et al., 2006).

In situ P mineralization (PMR) in soil was measured using a buried-bag technique (Chapin et al., 2003). Three paired soil cores were collected randomly in each of the three sites. One of the core samples from each pair was sealed in sterilized polythene bags after removing the coarse roots and larger fragments of organic debris, if any, to avoid nutrient immobilization during the incubation and reinserted to its respective depth. The other soil core was brought back to the laboratory, and composite samples were made for each depth in respect of each stand and sieved with a 2-mm mesh screen. After a gap of 1 month, the buried bags were retrieved on bimonthly intervals from each stand, and the soil samples were pooled according to the depth and analyzed for final concentrations of ammonium and nitrate. The P mineralization was calculated by subtracting the initial concentration from the final concentration.

The activities of acid phosphatases (ACP) and alkaline phosphatases (ALP) were assayed by colorimetric determination of p-nitrophenol released when soil was incubated with p-nitrophenyl phosphate in pH 6.5 or 11 buffers, respectively, for 1 h at 37°C (Tabatabai and Bremner, 1969). ACP and ALP activity was expressed as mg p-nitrophenol g^{-1} dry soil h^{-1}.

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Statistical analysis

One-way ANOVAs were conducted to test the effects of wetland degradation degree on plant communities, soil properties, soil P fraction and transformation. Multiple comparisons with LSD (least significant difference) were applied to examine differences among the degradation gradients. Pearson correlation analysis was performed to examine the relationships between soil P fractions and soil properties. Redundancy analysis (RDA) was used to explore the relationships between the different degradation gradients and plant and soil variables. Statistical procedures were conducted using SPSS version 24.0 (IBM, Armonk, USA) and Canoco version 5.0 (Microcomputer Power, Ithaca, USA).

Results

Plant communities and soil properties

The dominant plant community types in non-degradation alpine wetland of this area are *Carex muliensis*, *Tongoloa gracile*, *Gentianopsis paludosa*, *Caltha scaposa* (*Table 1*). These communities tended to shift to communities of *Kobresia setchwanensis*, *Poa pratensis*, and *Festuca sinensis* at moderate stages, and to *P. pratensis*, Anemone rivularis, and Argentina anserine at severe degradation. Plant coverage varied significantly among the four degradation stages, with the lowest values occurring in the SD. Living biomass, dead biomass and root biomass showed a downward trend with the development of wetland degradation. Degradation process significantly altered soil properties in the alpine wetland (*Table 1*). Soil pH increased in the MD and SD compared to the ND. The soil water content (SWC), soil organic carbon (SOC), and soil total nitrogen (STN) decreased with increasing degradation.

Table 1. *Characters of plant and soil at different degradation stages*

	ND	**LD**	**MD**	**SD**
Dominant species	*Carex muliensis, Tongoloa gracile, Gentianopsis paludosa, Caltha scapos*a	*Carex muliensis, Tongoloa gracilis, Kobresia setchwanensis, Thalictrum alpinum*	*Kobresia setchwanensis, Poa pratensi*s, *Festuca sinensis, Bistorta macrophylla*	*Poa pratensis, Anemone rivularis, Argentina anserine, Bistorta macrophylla*
Coverage (%)	94.2 ± 2.97a	78.9 ± 5.15b	62.3 ± 3.44c	47.7 ± 3.10d
Living biomass (g m^{-2})	376.47 ± 35.15a	341.31 ± 28.02a	309.40 ± 42.81a	195.77 ± 26.98b
Dead biomass (g m^{-2})	440.91 ± 28.33a	186.7 ± 20.27b	102.9 ± 12.57c	52.02 ± 5.29c
Root biomass (g m^{-2})	2745.23 ± 288.05a	2223.47 ± 191.77ab	1695.81 ± 211.99bc	1250.91 ± 50.97c
SWC (%)	189.05 ± 10.00a	166.50 ± 8.50b	96.26 ± 5.39c	48.72 ± 3.48d
pH	5.56 ± 0.04a	5.71 ± 0.06a	5.90 ± 0.06b	6.17 ± 0.05c
SOC (g kg^{-1})	265.08 ± 5.84a	192.49 ± 6.93b	124.84 ± 6.1c	64.36 ± 5.54d
TN (g kg^{-1})	18.45 ± 0.49a	12.05 ± 1.07b	8.72 ± 0.67c	6.07 ± 0.38d

ND, LD, MD and SD is non-degradation, light, moderate and severe degradation stages of the alpine wetland, respectively; SWC, Soil water content; SOC, Soil organic carbon; STN, Soil total nitrogen
Within rows, means ± SE. Different letters represent statistically significant ($p < 0.05$, one way ANOVA followed by LSD)

Plant P content

Plant P concentration tended to increase with the degradation gradient (*Fig. 1a, b, c*). Living plant P concentration increased by 27.17, 24.48, and 43.13% in the LD, MD, and HD, respectively compared with ND (*Fig. 1a*). Dead plant P concentration increased by 27.29 and 41.96% in the MD and HD, respectively compared with ND (*Fig. 1b*).

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Besides, there was no significant difference between LD and SD. Root P content of LD and SD raised by 29.63 and 22.50%, respectively compared with ND (*Fig. 1c*). However, there was no significant difference between LD and SD.

Plant P accumulation usually showed a decreasing trend with increasing levels of degradation (*Fig. 1 d, e, f*). Aboveground living biomass P accumulation at the HD sites was significantly lower than that at the ND, LD, and MD (*Fig. 1e*). Compared with ND, aboveground dead biomass P accumulation of LD, MD, and SD decreased by 51.59, 70.57, and 83.10%, respectively (*Fig. 1d*). Root P accumulation in ND site was not different to LD site, but was significantly higher than in MD and HD sites (*Fig. 1f*).

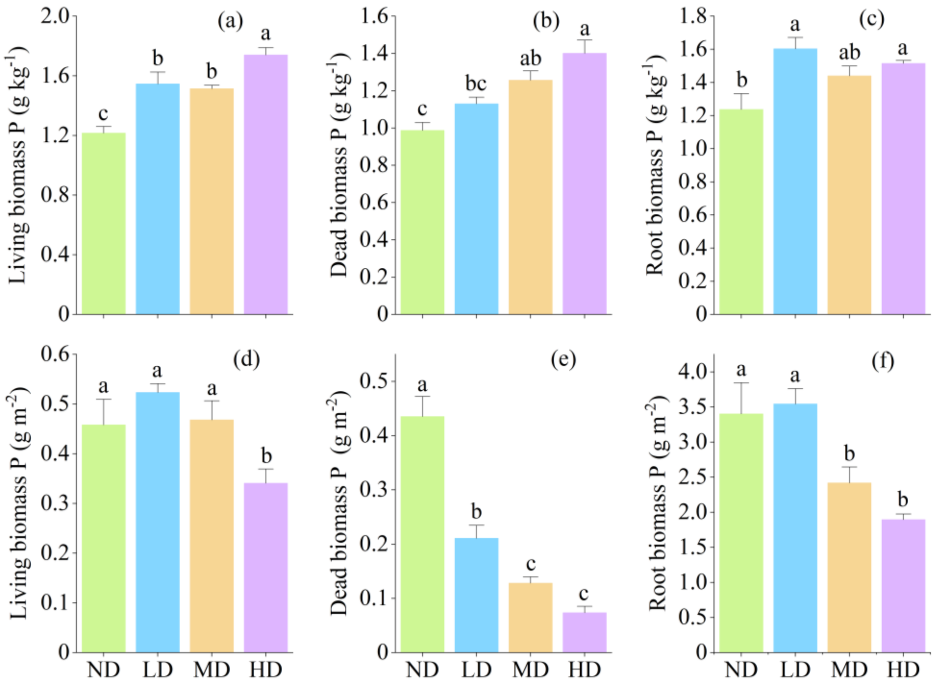

Figure 1. *The phosphorus (P) concentrations and P accumulation in aboveground living biomass, dead biomass and root biomass at the different degradation stages. Different letters indicate significant difference among sites ($p < 0.05$, one way ANOVA followed by LSD). ND is non-degradation and LD, MD, and SD are the light, moderate, and severe degradation stages of the alpine wetland, respectively*

Soil P content and fractions

Soil total P varied significantly among the four degradation stages, with the highest value of 2.47 g kg^{-1} in the LD site and the lowest value of 1.12 g kg^{-1} in the ND site (*Fig. 2a*). Available P was significantly higher in LD and MD sites than in ND and HD sites (*Fig. 2b*). Inorganic P in the ND site was 25.25 and 38.01% lower than that of LD, and MD, respectively but no significant difference was observed between ND and HD (*Fig. 2c*). Organic P in the ND site was 61.46, 57.95, and 48.96% lower than that of LD, MD, and HD site, respectively, but no significant difference occurred among three degradation degrees (*Fig. 2d*). Labile inorganic P in ND site was 37.72% lower than in the LD site, but 18.79% higher than in HD site (*Fig. 2e*). Labile organic P in the ND site showed no significant difference with three degradation sites, but it was significantly lower in HD site than in LD and MD sites (*Fig. 2f*).

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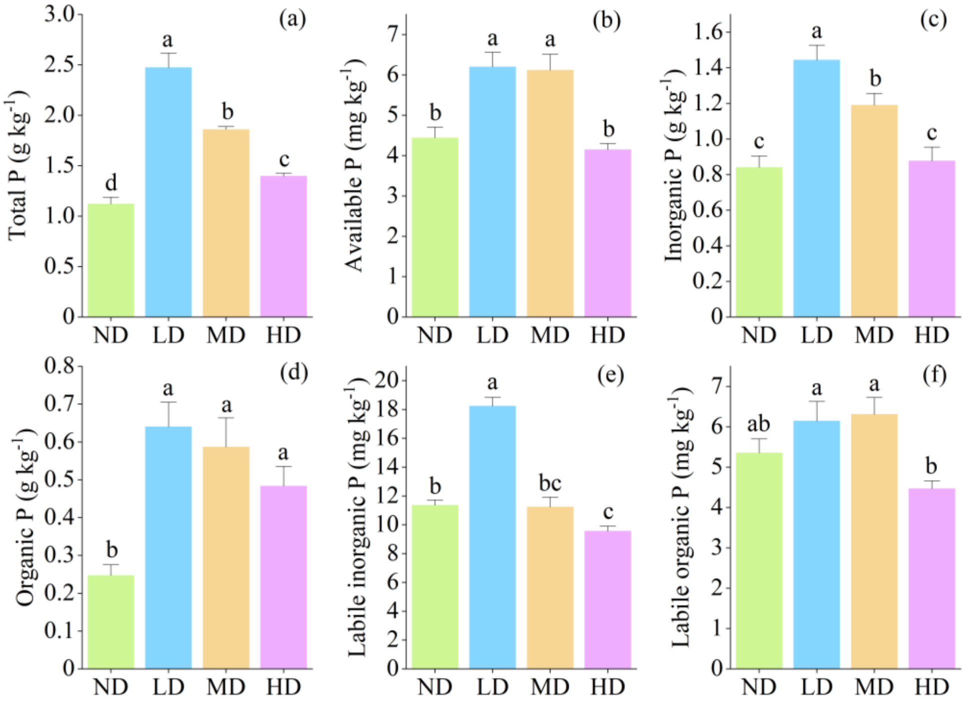

Figure 2. *The soil P content and fractions at the different degradation stages. Different letters indicate significant difference among stages (p < 0.05, one way ANOVA followed by LSD). ND is non-degradation and LD, MD and SD are the light, moderate and severe degradation stages of the alpine wetland, respectively*

Soil P mineralization and phosphatase activity

Soil P mineralization was increased firstly and decreased later on along the wetland degradation gradients. The trend showed the highest value of 1.21 mg P kg^{-1} d^{-1} in the LD site and the lowest value of 0.66 mg P kg^{-1} d^{-1} in the SD site (*Fig. 3a*). Soil ACP in the ND was not different to LD site, but was 23.40, and 42.51%, lower than in MD, and HD sites, respectively (*Fig. 3b*). Soil ALP in the MD area was significantly higher than that in other sites, reaching 3.23 mol PNP kg^{-1} h^{-1}, besides, there was no significant difference between ND and MD (*Fig. 3c*).

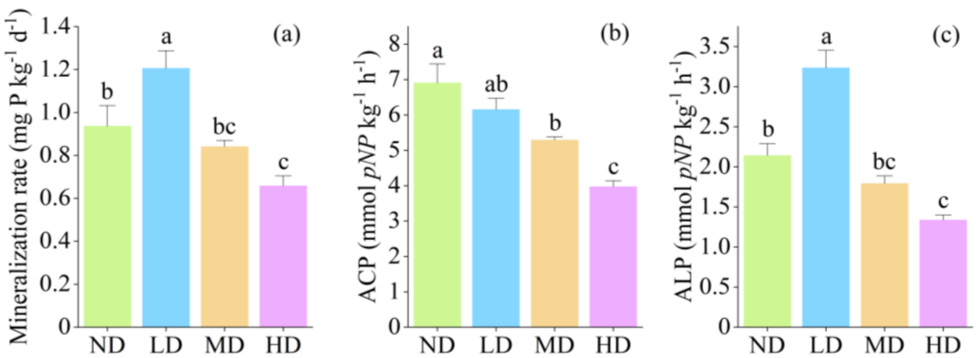

Figure 3. *The soil P mineralization and phosphatase activities at the different degradation stages. Different letters indicate significant difference among stages (p < 0.05, one way ANOVA followed by LSD). ND is non-degradation and LD, MD, and SD are the light, moderate and severe degradation stages of the alpine wetland, respectively*

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Relationship among plant biomass, P content, and soil variables

The Pearson correlation analysis showed that aboveground living biomass was significantly and positively correlated with SOC, TN, LOP, PMR, and ACP (*Fig. 4*). The dead biomass and root biomass were significantly related to SOC, TN, and ACP (*Fig. 4*). The living biomass P was positively related to soil LIP, LOP, PMR, and ALP (*Fig. 4*). The dead biomass P was positively correlated with SOC, TN, and ACP, but negatively with soil OP. The root P was significantly and positively correlated with SOC, TN, LIP, PMR, ACP, and ALP. The plant P component was negatively correlated to the soil pH (*Fig. 4*). Soil available P correlated significantly and positively with IP, LIP, and LOP (*Fig. 4*).

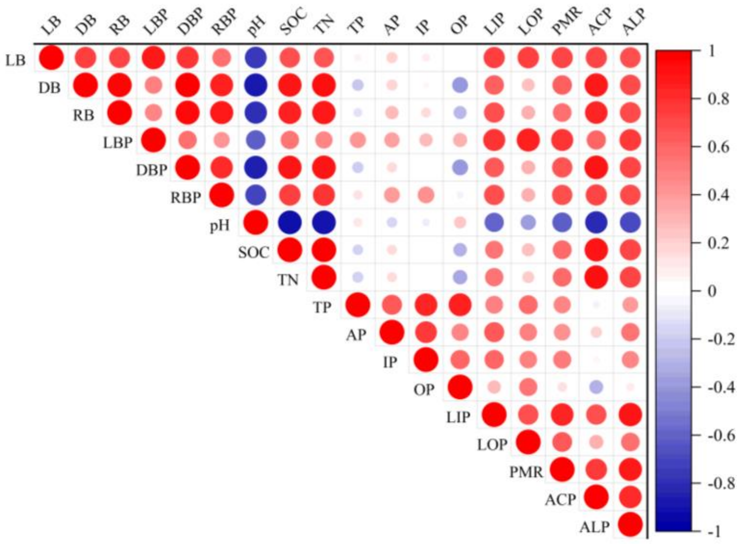

Figure 4. *Heatmap of the correlations between plant P and soil properties. A blue square indicates that the correlation coefficient is positive, and a red circle indicates that the correlation coefficient is negative. The correlation coefficient decreased as the color became lighter and the size became smaller. LB, living biomass; DB, dead biomass; RB, root biomass; LBP, living biomass P; DBP, dead biomass P; RBP, root biomass P; SOC, soil organic carbon; TN, total nitrogen; IP, inorganic P; OP, organic P; LIP, labile inorganic P; LOP, labile organic P; PMR, P mineralization; ACP, acid phosphatases, ALP, alkaline phosphatases*

The RDA showed that the first and two canonical axes explained 57.3 and 27.3% of the total variation in the plant biomass and P storage, respectively (*Fig. 5*). SOC, TN, ACP, PMR, ALP, LIP, LOP, and pH were the main factors affecting the change in plant biomass and P storage in the different degradation gradients, and they were highly correlated. The LD had greater effects than M on shoot P, LIP, LOP, and AP, while MD had the highest TP and OP. Compared with LD, HD had negative effects on shoot biomass and root P, but had positive effect on pH.

Discussion

In the recent years, the degradation of alpine wetlands on the Tibetan Plateau has been obvious, and the ecosystem has undergone reverse succession, resulting in different degrees of changes in plant communities and soil properties in the region

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(Yang et al., 2022; Ren et al., 2013). In this study, the status of *C. muliensis*, and *T. gracile*, the original community species of alpine wetland, gradually weakened with the degradation of swampy wetland. On the other hand, the dry mesophytes such as *K. setchwanensis* and *P. pratensis* evolved into the main component species of the community. The result was consistent with the trend of the plant community structure changes in the degradation of the Shouqu wetland of the Yellow River and the Maqin wetland of the Qinghai Province (Ren et al., 2013; Hou et al., 2009). The reason for this result is that after the degradation of the wetland, the soil moisture was significantly reduced, the grassland tended to be arid, and the plant habitat changed significantly, which caused the plant community succession (Hou et al., 2009; Gao et al., 2011).

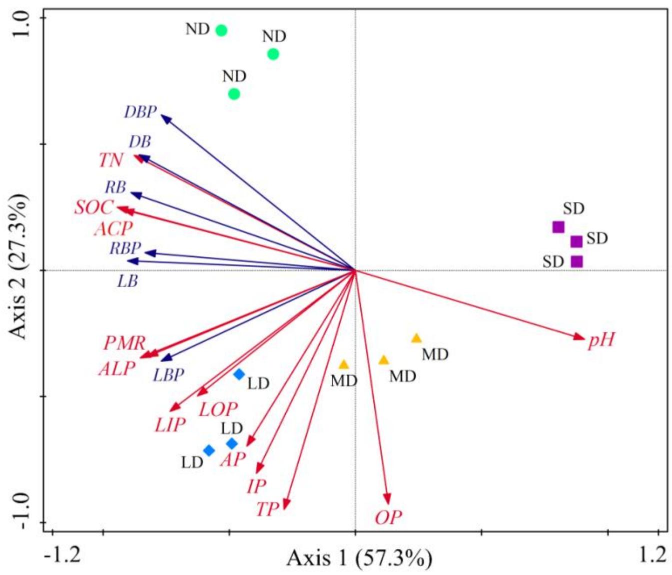

Figure 5. *Ordination plots of the redundancy analysis (RDA) for plant and soil variables across all stages. ND, non-degradation; LD, light degradation; MD, moderate degradation; SD, severe degradation. LB, living biomass; DB, dead biomass; RB, root biomass; LBP, living biomass P; DBP, dead biomass P; RBP, root biomass P; SOC, soil organic carbon; TN, total nitrogen; IP, inorganic P; OP, organic P; LIP, labile inorganic P; LOP, labile organic P; PMR, P mineralization; ACP, acid phosphatases, ALP, alkaline phosphatases*

The turnover of plant species in alpine wetlands with the time of degradation succession not only affects the structural characteristics of plant communities, but also influences the changes in plant productivity (Yang et al., 2022). In the early stage of the degradation succession, the plant community was dominated by sedge plants such as the clonal *C. muliensis*. At this stage the weed species and biomass were small, the vegetation cover was high, and its accumulated aboveground biomass was also high. With the reduction of water accumulation in marshy wetlands, the basic withdrawal of native plants and the increase of surface bald patches and bare ground resulted. During which the total biomass value of the community was the lowest, while the proportion of forage biomass of grasses and sedge family were decreased, the production performance was also decreased accordingly. The degradation of wetland ecosystem function was accelerated by the change in plant community structure and loss of species diversity brought about by the wetland degradation process.

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Soil pH increased significantly with the degree of degradation of alpine wetlands, indicating that degraded wetland soils showed an acidic-neutral-alkaline trend. The finding was consistent with the study in Maqin County, Tibetan Plateau (Ren et al., 2013). The increase in pH with the degree of degradation, probably resulted because of the reduction of vegetation caused by degradation and the destruction of organic acids in the soils by intense solar radiation (Gao et al., 2011). Soil organic C and total N content decreased significantly with the increase of wetland degradation, which was consistent with the results of previous studies (Yang et al., 2022; Ren et al., 2013; Gao et al., 2011). In a stable ecosystem, soil organic carbon content is determined by the balance between plant organic matter inputs and respiratory losses by soil anaerobic microorganisms (Yang et al., 2022). When ecosystem retreat occurs, vegetation productivity reduces the amount of soil organic matter input and accelerates the rate of soil organic matter decomposition, resulting in a significant loss of soil organic matter (Yang et al., 2022). At the same time, the deterioration of vegetation cover conditions will further exacerbate the effects of water and wind erosion and continue to strip the soil, resulting in the loss of organic C (Gao et al., 2011).

The P concentration in shoot and root biomass tended to increase with wetland degradation, which may be due to the dilution of P in the plant as a result of plant growth (Moeneclaey et al., 2002). In the undegraded wetland, plant growth was not affected, and the P content in the plant body was diluted by the normal growth process. However, in the degraded wetland, the plant growth slowed down, and the P content in the plant body was relatively higher. In addition, plant nutrients were derived from the elements in the soil, and the deterioration of the nutrient status of the degraded wetland's soil organic C and N affected the uptake of plant P (Yang et al., 2022; Gao et al., 2011). The P content of litter biomass had a similar pattern to the P concentration of aboveground living biomass and this result suggested that P concentration of the little biomass was subjected to that of the live aboveground biomass. P content in plant components showed similar patterns as their biomass under different degradation degree, and no changes occurred in P allocation in aboveground and belowground. Those results suggested that changes in above- and below-ground plant parts are synergistic during the processes of wetland degradation.

On the degradation gradient of alpine wetlands, soil total P content in all showed an increasing trend followed by a decreasing status. But the P content was still higher than that of the marsh stage i.e., the most serious stage of degradation. It indicated that the degradation process of alpine wetlands is a process of soil P aggregation and the finding is consistent with the results of the study on the degradation of the Mauwusu wetland (He et al., 2019). It happened mainly due to the long-term flooding of the native marsh wetland, which promotes the transfer of phosphorus from the wetland sediments to the overlying water body under low redox potential conditions (Reddy et al., 1999; Yang et al., 2016). The process increased the amount of P loss with water and thus reducing the accumulation of P in the soil. During the subsequent stages of degradation of the wetland, the soil total P content decreased, mainly due to the reduction of plant biomass and mineralization of organic matter (He et al., 2019). In addition, changes in iron and aluminum oxides content due to changes in moisture conditions may be another important reason for changes in soil total P content (Reddy et al., 1999; Jan et al., 2015). It has been shown that changes in soil moisture can change the redox conditions of the soil, which in turn affects the content of iron and aluminum oxides in the soil, which are directly related to the process of adsorption and release of soil P (Jan et al., 2015).

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The increase in soil P accumulation after the degradation of the Ruoergai marsh wetland can be regarded as a result of the increase in the content of organic and inorganic P, and the inorganic P served as the main form of soil total P. The highest content of inorganic P and active inorganic P was found in the mildly degraded wetland. It resulted because the wetland was mildly degraded with improved soil aeration, root metabolism, and enhanced microbial activities (Yang et al., 2022; Gao et al., 2011). Therefore, P mineralization increased, resulting in different degrees of activation of soil P and a significant increase in active organic P (Chapin et al., 2003; Li et al., 2021). When the wetland degraded further, vegetation growth restricted, soil organic matter and phosphatase activity reduced significantly. All these led to a decrease in the amount of inorganic P activation, inhibition of organic P decomposition, and a decrease in soil P effectiveness. Different P forms have a direct impact on soil P effectiveness, and phosphatase is the main driving force to regulate the transformation of different forms of P in soil (Song et al., 2007; Wang et al., 2006). In this process, the acid phosphatase mainly originates from plant roots and microorganisms, and alkaline phosphatase mainly from microorganisms (Li et al., 2021). In this study, ALP was significantly and positively correlated with soil effective P, indicating that soil ALP in alpine wetlands is an important driving factor for P turnover.

Conclusions

Degradation of alpine wetlands resulted in the succession of plant communities from wet to dry plants, with a significant decrease in plant cover and biomass. During wetland degradation, soil organic C and total N showed a linear decrease, but light and moderate degradation improved soil total P accumulation and increased inorganic P content by promoting the mineralization rate under the action of phosphatase, and improved the effectiveness of soil P. However, when the wetland undergoes heavy degradation, the increase of total and effective P is limited by the combined effects of soil moisture and vegetation communities. Under this condition, the supply capacity of soil P decreases, and the risk of ecological deterioration of the wetland increases. This paper emphasizes the nonlinear changes of soil P and its composition during wetland degradation, and further suggests that the restoration strategy of heavily degraded wetlands should consider the enhancement of soil P effectiveness, especially the supplementation of inorganic P.

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