Dynamics of Soil Respiration in Alpine Wetland Meadows Exposed to Different Levels of Degradation in the Qinghai-Tibet Plateau, China

Zhongfei Li1,2, Jixi Gao2, Linqin Wen1, Changxin Zou2, Chaoyang Feng3,4, Daiqing Li3,4 & Delin Xu2

The effects of degradation of alpine wetland meadow on soil respiration (Rs) and the sensitivity of Rs to temperature (Q10) were measured in the Napa Lake region of Shangri-La on the southeastern edge of the Qinghai-Tibet Plateau. Rs was measured for 24 h during each of three different stages of the growing season on four different degraded levels. The results showed: (1) peak Rs occurred at around 5:00 p.m., regardless of the degree of degradation and growing season stage, with the maximum Rs reaching 10.05 μmol·m⁻²·s⁻¹ in non-degraded meadows rather than other meadows; (2) the daily mean Rs value was 7.14–7.86 μmol·m⁻²·s⁻¹ during the mid growing season in non-degraded meadows, and declined by 48.4–62.6% when degradation increased to the severely degraded level; (3) Q10 ranged from 7.1–11.3 in non-degraded meadows during the mid growing season, 5.5–8.0 and 6.2–8.2 during the early and late growing seasons, respectively, and show a decline of about 50% from the non-degraded meadows to severely degraded meadows; (4) Rs was correlated significantly with soil temperature at a depth of 0–5 cm (p < 0.05) on the diurnal scale, but not at the seasonal scale; (5) significant correlations were found between Rs and soil organic carbon (SOC), between biomass and SOC, and between Q10 and Rs (p < 0.05), which indicates that biomass and SOC potentially impact Q10. The results suggest that vegetation degradation impact both Rs and Q10 significantly. Also, we speculated that Q10 of alpine wetland meadow is probable greater at the boundary region than inner region of the Qinghai-Tibet Plateau, and should be a more sensitive indicator in the studying of climate change in this zone.

Research indicates that atmospheric CO₂ concentrations rose from 280 ppm in 1975 to 397 ppm in 20141, and will potentially rise to 500–1000 ppm by 2100 if no corrective actions are taken2,3. Atmospheric CO₂ concentrations are strongly influenced by carbon flux in terrestrial ecosystems, especially by soil respiration (Rs) processes, which can emit ~120 Pg of carbon to the atmosphere per year4. In terrestrial ecosystems, the amount of carbon emitted from Rs processes is second only to the amount of carbon fixed by gross primary productivity (GPP) and is even more than the carbon uptake by net primary productivity (NPP) in certain situations5–9.

Rs is a key component of carbon flux in the global carbon cycle and a potential indicator of ecosystem metabolism10,11. It can also be used to estimate belowground carbon allocation12, and to reveal the processes and mechanisms of carbon sources and sinks on regional and global scales. More precisely, Rs can be used to predict future atmospheric CO₂ concentrations and the degree and rate of climate change4,13. However, due to the high temporal and spatial heterogeneity of Rs, it can only be accurately measured directly within each specific location, which makes it difficult to simulate, predict, and assess the spatial and temporal dynamics of Rs at global and regional scales14, and to identify how these dynamics contribute to climate change15,16. Therefore, quantitative field measurement for Rs in various ecosystem types is still urgently needed in research of global carbon cycle, which will contribute to reducing uncertainty when quantifying ecosystem carbon sequestration17.
Wetland soils, which constitute only 2–3% of global land area, store a disproportionate amount of global soil carbon (18–30% of the total 1550 Pg of soil carbon)\(^1^8\)\(^,\)\(^1^9\). However, data on soil carbon dynamics in wetland regions is scarce\(^2^0\). For example, there are only 135 records for wetlands among a total of 3821 records in the global Rs database (SRDB version 20100517)\(^1^6\),\(^2^0\). China contains about 4% wetlands, making it one of the richest countries in terms of wetland resources in the world\(^2^1\). Wetlands are typically carbon sinks\(^2^2\)–\(^2^5\), because the anaerobic environment, high productivity, and low soil temperature (Ts) can tend to reduce decomposition and promote peat formation\(^2^6\)–\(^3^2\).

Some studies identify the alpine wetland meadows of the Qinghai-Tibetan Plateau (QTP) in China as a huge organic carbon sink that is highly sensitive to global climate change\(^3^3\),\(^3^4\). However, other studies classify the QTP wetland meadows as a carbon source\(^3^5\),\(^3^6\). It is therefore important to resolve this discrepancy by examining the influence of different types or conditions of aboveground vegetation on soil carbon emission\(^3^7\). There is an urgent need to understand the carbon exchange processes occurring in alpine wetland meadows\(^3^4\),\(^3^8\).

The Shangri-La region, located in northwestern Yunnan Province at the eastern edge of the QTP, lies in the Hengduan Mountains\(^3^9\),\(^4^0\). It contains rich biodiversity and serves as an important ecosystem service spillover region. Numerous plateau lakes and alpine wetland ecosystems are distributed throughout the region, with characteristics typical of a low-latitude and high-altitude geographical environment. As global climate change and human adaptation progress, however, these alpine wetland ecosystems face unprecedented threats, including drainage transformation, reclamation, tourism development, and shortage of water resources. A large number of seasonally-flooded alpine swamp meadows have been experiencing changes in ecosystem services, which has turned these swamp meadows into alpine wetland meadows or alpine meadows and caused shifts in community composition and structure, and the physical and chemical properties of soil and environmental conditions. Constant disturbances from human activities, such as overgrazing of livestock and trampling by tourists, have caused a great number of wetland meadows to further degrade. Ultimately, alpine wetland meadow degradation leads to changes in carbon budgets and the balance of ecosystems\(^2^9\),\(^4^1\),\(^4^2\).

The boundary region of the QTP may exhibit a more sensitive response to climate change than other regions\(^4^3\). In the past few decades, the QTP has experienced more rapid warming than other regions of the world\(^4^4\)–\(^4^6\). Many studies on carbon flux in the QTP have been conducted\(^3^7\),\(^4^7\)–\(^4^9\), including studies in alpine meadows, and wetland meadows\(^6\)^,\(^5^0\)–\(^5^3\), but almost all of these studies have been conducted in the interior region of the QTP instead of in the boundary region. Research in the Shangri-La region on the southeastern edge of the QTP is almost nonexistent. The dual effects of human activities and climate change on carbon flux in the natural ecosystems of this region are still unknown.

For this study, we selected alpine wetland meadow, which is one of three grassland types (including alpine meadow, alpine shrubland meadow, and alpine wetland meadow) in QTP, and is perennially exposed to water, in the Napa Lake region of Shangri-La, located on the southeastern edge of the QTP. Within the alpine wetland meadows, we identified four levels of degradation severity, based on fencing and grazing, tourism trampling, vegetation coverage, and aboveground biomass. Rs was measured for 24 h within plots of different degradation severities during the early, middle, and late stages of the growing seasons in 2014 and 2015. The objectives of this study were: (1) to understand the dynamic and variable mechanism of Rs in differently degraded alpine wetland meadows at diurnal, seasonal, and inter-annual time scales; (2) to reveal the effects of vegetation degradation on Rs in alpine wetland meadows; This paper provides a better understanding of the effects of human activities on carbon cycling between land and atmosphere in the QTP region.

**Results**

**Thirty-year changes in temperature and precipitation in the study area.** Figure 1 shows that the average annual temperature in the study region rose from 5.9 °C in 1981 to 7.5 °C in 2015, for a total increase of 1.6 °C. The average temperature increase between 1990 and 2000 was 0.37 °C greater than the average temperature increase between 1981–1990, and the average increase between 2000 and 2010 was 0.60 °C greater than that between 1990 and 2000, suggesting that the size of the temperature increase has grown over time.
Precipitation presents a decreasing trend ($p < 0.01$) (Fig. 1). The precipitation decline mainly occurred after 2005. Average annual precipitation between 2006 and 2015 was only 542.4 mm, which is much lower than the annual averages between 1981 and 1990 (628.5 mm) and between 1990 and 2000 (696.6 mm) (Fig. 1).

### Vegetation conditions in alpine wetland meadows impacted by different levels of degradation

Table 1 shows that vegetation coverage, aboveground biomass, and LAI were significantly lower in the SDM than in the NDM ($p < 0.05$); vegetation coverage and biomass were 50% lower and LAI was 80% lower. Vegetation coverage and aboveground biomass were about 40% and 75% lower in the SDM than the NDM. This suggests that alpine wetland meadow degradation results in a significant decrease in the condition of aboveground vegetation ($p < 0.05$).

### SOC in alpine wetland meadows impacted by different levels of degradation

Table 2 shows that the carbon content of vegetation in alpine wetland meadows decreased significantly ($p < 0.05$) with increasing degradation. For example, carbon content was approximately 15.8% lower in the SDM than in the NDM. SOC content in the 0–30 cm soil layer declined significantly between the NDM and SDM levels of degradation ($p < 0.05$). Specifically, SOC declined by 56.4% in the 0–10 cm soil layer, 61.2% in the 10–20 cm layer, and 64.1% in the 20–30 cm layer. But there was no significant difference in SOC content ($p > 0.05$) between the NDM and LDM levels in the 30–40 cm soil layer, nor among any of the degradation levels in the 40–50 cm soil layer ($p > 0.05$) (Table 2).

### Diurnal and seasonal Rs variation within different levels of degradation in 2014 and 2015

Figures 2 and 4 show the diurnal and seasonal variations in Rs and Ts in plots impacted by different levels of degradation in 2014 and 2015. Rs in all plots showed a single peak in the diurnal analysis, which occurred at around 5:00 pm. Beginning at 7:00 a.m., Rs fluctuated in an increasing direction from 7:00 am until the peak, and then fluctuated in a decreasing direction until 6:30 am the next morning. Peak Ts values appeared between 3:00 p.m.–8:00 p.m. in every plot (Figs 2 and 3).

As for seasonal variations, Ts were highest during the MGS and were basically equivalent during the EGS and LGS at all levels of degradation in both 2014 and 2015 (Figs 2 and 3). Diurnal and seasonal variations in Rs and Ts within the different degraded plots were similar between 2014 and 2015.

### Variation in daily peak Rs at different levels of degradation

Figure 4 shows that peak Rs decreased significantly ($p < 0.05$) with increasing degradation. Peak Rs ranged from 4.64–10.05 μmol·m$^{-2}$·s$^{-1}$ in the NDM and LDM plots, but only 1.75–4.93 μmol·m$^{-2}$·s$^{-1}$ in the MDM and SDM plots. In 2015, peak Rs was 5.37 μmol·m$^{-2}$·s$^{-1}$ during the EGS, 8.78 μmol·m$^{-2}$·s$^{-1}$ during the MGS, and 4.74 μmol·m$^{-2}$·s$^{-1}$ during the LGS in

---

### Tables

**Table 1. Vegetation conditions in alpine wetland meadows impacted by different levels of degradation.** Note: (1) The data is mean value ± SD. Different lowercase letters within a line indicate a significant difference between mean values ($p < 0.05$) for different levels of degradation. (2) NDM, LDM, MDM, and SDM represent non-degraded meadow, lightly-degraded meadow, moderately-degraded meadow and severely-degraded meadow respectively.

| Degraded Levels | Vegetation Coverage (%) | Aboveground Biomass (g·m$^{-2}$) | LAI | Dominant Species | Human Activity          |
|-----------------|-------------------------|---------------------------------|-----|-----------------|-------------------------|
| NDM             | 95.0 ± 1.6a             | 304.8 ± 14.9a                   | 2.4 ± 0.2a | Blysmus sinocompressus, Carex muriensis | Fenced over 20 years, no grazed, reaped and tourism disturbance |
| LDM             | 84.3 ± 3.3b             | 246.2 ± 3.0b                    | 1.8 ± 0.1b | Blysmus sinocompressus, Carex muriensis | Fenced over 10 years, reaped per year, no grazed and tourism disturbance |
| MDM             | 66.7 ± 2.4c             | 101.7 ± 5.7c                   | 0.9 ± 0.1c | Potentilla anserina, Pedicularis longiflora | Grazed, but no fenced and tourism disturbance |
| SDM             | 50.0 ± 4.1d             | 78.5 ± 4.5d                    | 0.4 ± 0.1d | Potentilla anserina | Grazed and tourism disturbance, but no fenced |

**Table 2. SOC of alpine wetland meadows impacted by different levels of degradation.** Note: (1) The data is mean value ± SD. Different lowercase letters within a line indicate a significant difference between mean values ($p < 0.05$) for different levels of degradation. Capital letters within a row indicate a significant difference between soil layers ($p < 0.05$). (2) NDM, LDM, MDM, and SDM represent non-degraded meadow, lightly-degraded meadow, moderately-degraded meadow and severely-degraded meadow respectively. (3) SOC is soil organ carbon.

| Degraded Levels | Carbon Content of Vegetation (g·kg$^{-1}$) | SOC in Different Soil Layer (g·kg$^{-1}$) |
|-----------------|--------------------------------------------|------------------------------------------|
|                 |                                            | 0–10 cm | 10–20 cm | 20–30 cm | 30–40 cm | 40–50 cm |
| NDM             | 466.9 ± 12.1a                              | 47.0 ± 0.9aA   | 25.8 ± 2.0aB | 18.1 ± 0.4aC | 14.7 ± 1.1aC | 7.8 ± 0.7aD |
| LDM             | 457.7 ± 7.7ab                              | 33.7 ± 0.9aA   | 20.1 ± 0.6aB | 14.7 ± 0.5aC | 13.7 ± 0.3aC | 6.4 ± 0.5aB |
| MDM             | 443.4 ± 6.4b                              | 25.6 ± 1.5aA   | 17.0 ± 0.8aB | 11.4 ± 0.4aC | 6.9 ± 0.4aB | 6.9 ± 0.6aB |
| SDM             | 392.9 ± 7.6c                              | 20.5 ± 0.6aA   | 10.0 ± 0.6aC | 6.5 ± 0.9aC | 5.0 ± 0.3aB | 6.8 ± 0.6aC |
During the same stages of the growing season in the SDM, peak Rs was lower by 39.7% during the EGS, 57.6% during the MGS, and 63.1% during the LGS. Peak Rs was highest during the MGS. Peak Rs was lower, but similar, during the EGS and LGS. Peak Rs values during the MGS were about 44.0% higher than during the other two growing season stages in the NDM and about 35.0% higher in the LDM, but only 26.0% higher in the MDM and 24.2% higher in the SDM (Fig. 4).

**Daily mean Rs value at different levels of degradation.** Figure 5 shows the daily mean Rs values at different levels of degradation. Overall, the values decreased significantly as the level of degradation increased \( (p < 0.05) \). The daily mean Rs value reached 4.24 \( \mu\text{mol} \cdot \text{m}^{-2} \cdot \text{s}^{-1} \) during the EGS, 7.86 \( \mu\text{mol} \cdot \text{m}^{-2} \cdot \text{s}^{-1} \) during the MGS, and 5.22 \( \mu\text{mol} \cdot \text{m}^{-2} \cdot \text{s}^{-1} \) during the LGS in the NDM in 2014. These values were 48.4%, 62.6%, and 53.2% lower, respectively, in the SDM. Daily mean Rs values in the NDM in 2015 were 4.39 \( \mu\text{mol} \cdot \text{m}^{-2} \cdot \text{s}^{-1} \) during the EGS, 7.14 \( \mu\text{mol} \cdot \text{m}^{-2} \cdot \text{s}^{-1} \) during the MGS, and 4.09 \( \mu\text{mol} \cdot \text{m}^{-2} \cdot \text{s}^{-1} \) during the LGS, which, similar to 2014, were higher than the daily mean Rs values in the SDM by 55.8%, 61.2%, and 62.8%, respectively.

The daily mean Rs value during the MGS was approximately 40.0% higher than during the other two growing season stages in the NDM, about 28.6% in the LDM, but only 26.0% higher in the MDM and 24.2% higher in the SDM (Fig. 4).

**Correlation of Rs and Ts.** Figures 6 and 7 show the power exponential curve relationship between Rs and Ts at the 0–5 cm soil depth within different levels of meadow degradation and during different stages of the growing season in 2014 and 2015. Almost all correlation coefficients \( (R^2) \) were above 0.5, and most of them were above 0.6. All power exponents passed the significance test \( (p < 0.01) \).

**3.8. \( Q_{10} \) values for different levels of degradation.** Figure 8 shows the variation characteristics of \( Q_{10} \) in different levels of degradation during different stages of the growing season. The \( Q_{10} \) in the NDM reached 5.5, 7.1, and 6.2 during the EGS, MGS, and LGS, respectively, in 2014, but decreased from the NDM levels by more than 30%, 40%, and 50% in the LDM, MDM, and SDM, respectively. \( Q_{10} \) was 8.0, 11.3, and 8.2 in the NDM during the EGS, MGS, and LGS, respectively, in 2015; these values were slightly higher than in 2014, but the variation between the two years was very similar.

The \( Q_{10} \) values during the MSG were higher than during the ESG and LSG. There was no significant difference between the ESG and LSG \( Q_{10} \) values.
Correlations among vegetation biomass, SOC, Rs, and Q10. Table 3 shows the correlations among aboveground biomass, SOC, daily mean Rs value, and Q10 in alpine wet meadows impacted by four different levels of degradation. Aboveground biomass correlated significantly with SOC at the 0–10 cm and 30–40 cm soil depths (p < 0.05), but not at other soil depth (p > 0.05), and with Q10 (p < 0.05).

SOC at the 0–10 cm soil depth correlated significantly with Rs and Q10 (p < 0.05). At the 10–20 cm soil depth, SOC and Rs correlated significantly (p < 0.05), as did Rs and Q10 (p < 0.05).
Figure 5. Daily mean Rs value at different levels of degradation in alpine wetland meadows in 2014 and 2015. Note: Different lowercase letters indicate significant differences between daily mean Rs values ($p < 0.05$) at different levels of degradation. Capital letters indicate significant differences between growing season stages in the same year. Bars indicate SE of mean.

Figure 6. Exponential correlation of Rs and Ts at the 0–5 cm soil depth in 2014.
Figure 7. Exponential correlation of Rs and Ts at the 0–5 cm soil depth in 2015.

Figure 8. The $Q_{10}$ values for meadows with different levels of degradation in 2014 and 2015.
Discussion

The impacts of degradation on Rs in a Napa Lake alpine wetland meadow. Aboveground vegetation degradation has an important modulating factor on Rs\(^{66-58}\). In this study, the positive correlation between Rs (in 2015) and aboveground biomass was significant (Table 2), and the positive correlation between Rs and SOC was significant (Table 3). Moreover, the SOC of the top soil layers synchronously (Tables 2, 3). These direct and indirect effects can ultimately decrease the Rs rate.

Studies have shown that aboveground biomass is an important modulating factor of Rs\(^{66-58}\). In this study, the positive correlation between Rs (in 2015) and aboveground biomass was significant (Table 3), and the positive correlation between Rs and SOC was significant (Table 2). Moreover, the SOC of the top soil layers correlated directly with aboveground vegetation biomass (Table 3). Degradation decreases aboveground biomass, which in turn reduces the activities of biological processes of roots and soil\(^{59,60}\), and SOC content was observed to decrease synchronously (Tables 2, 3). These direct and indirect effects can ultimately decrease the Rs rate.

Aboveground wetland meadows that are perennially exposed to water are typical ecological systems in the Napa Lake region of Shangri-La. Pervasive and long-term disturbances from grazing have caused most of these alpine wetland meadows to become degraded. Grazing has heavily impacted Rs rates\(^{61}\) by affecting soil nutrients\(^{62,63}\), Ts\(^{64,65}\), and aboveground vegetation\(^{66,67}\). Meanwhile, other study conducted in this region found that grazing did not significantly soil respiration\(^{68}\). Different findings maybe leaded by different vegetation types here. Anyhow, in recent years, frequent trampling from tourism activities in the region has become a more important factor leading to severe degradation of alpine wetland meadows. Together, these human activities have caused serious degradation of alpine wetland meadows, which has disrupted regional carbon balances.

Sensitivity of Rs to temperature, and variation of Q\(_{10}\). Ts is one of the most important factors governing Rs processes on different spatial-temporal scales\(^{69-71}\), but much uncertainty remains regarding the influence of other factors on Rs\(^{72-80}\). In our study, Rs displayed a significant exponential correlation with Ts on the scale of diurnal variation (Figs 6 and 7), but no significant correlation at the seasonal scale. In contrast, seasonal fluctuations in Rs correlated consistently with seasonal fluctuations in vegetation biomass at every level of degradation in this study (Table 1, Fig. 5). Thus, some researchers have concluded that temperature does not adequately account for all Rs variations\(^{72,76,81}\), and that vegetation is also key factor influencing the seasonal scale\(^{72,82,83}\).

The Q\(_{10}\) value, which is the amount that Rs increases with each 10 °C rise in temperature, has commonly been used to assess the sensitivity of Rs to temperature across a variety of ecosystem types and climatic zones\(^{72,84-87}\). In this study, we found that the Q\(_{10}\) of Rs declined as degradation severity increased in an alpine wetland meadow system (Fig. 8), and that the Q\(_{10}\) showed a significant direct correlation with SOC (p < 0.05) and with aboveground biomass (Table 3). These results, which appeared in different seasons and years, are similar to those of other works\(^{71,88}\) who studied the alpine meadows of Haibei in the QTP. These results suggest that vegetation degradation can directly reduce the sensitivity of Rs to Ts in the alpine wetland meadows of Napa Lake in Shangri-La.

More and more evidence shows that Q\(_{10}\) represents a combination of several influencing factors\(^{89,90}\), including biotic and abiotic factors\(^{90,91-93}\). In this study, the Q\(_{10}\) value was higher during the MGS than during the EGS or LGS (Fig. 8), which is consistent with the seasonal dynamics of aboveground biomass and Rs (Table 1, Fig. 5). A similar seasonal variation pattern in Q\(_{10}\) was observed by\(^{87}\) in their study of an alpine meadow in Haibei, QTP.

Overall, whether on a time scale of different seasons or different severities of vegetation degradation, the aboveground vegetation condition exerts a significant and decisive influence on Q\(_{10}\) in this study.

Values of Rs and Q\(_{10}\) based on transverse comparison with other studies in Shangri-La. Rs is the rate of CO\(_2\) release from the soil to the atmosphere, and Q\(_{10}\) is the sensitivity of Rs to temperature changes. Comparatively high Rs rates indicate relatively high vegetation activity\(^{71,81}\), decomposition rates of soil organic matter\(^{85,89}\), SOC content\(^{72}\), soil microbial biomass\(^{57,84}\), soil microbial activity\(^{96,97}\), and soil moisture\(^{75,76,80,98}\), etc.

### Table 3. Correlations among aboveground biomass, SOC, Rs, and Q\(_{10}\) in 2014 and 2015.

| Biomass | SOC1 | SOC2 | SOC3 | SOC4 | SOC5 | Rs1 | Rs2 | Q\(_{101}\) | Q\(_{102}\) |
|--------|-----|-----|-----|-----|-----|-----|-----|--------|--------|
| Biomass | 1   |     |     |     |     |     |     |        |        |
| SOC1   | 0.96* | 1   |     |     |     |     |     |        |        |
| SOC2   | 0.92 | 0.96* | 1   |     |     |     |     |        |        |
| SOC3   | 0.94 | 0.96* | 0.99** | 1   |     |     |     |        |        |
| SOC4   | 0.99* | 0.92 | 0.91 | 0.94 | 1   |     |     |        |        |
| SOC5   | 0.45 | 0.68 | 0.57 | 0.51 | 0.33 | 1   |     |        |        |
| Rs1    | 0.90 | 0.99** | 0.94* | 0.95 | 0.95 | 0.62 | 1   |        |        |
| Rs2    | 0.99* | 0.99** | 0.96* | 0.97* | 0.95* | 0.61 | 0.99** | 1   |        |
| Q\(_{101}\) | 0.98* | 0.98* | 0.97* | 0.98* | 0.97* | 0.83 | 0.79 | 0.97* | 0.99* |
| Q\(_{102}\) | 0.98* | 0.98* | 0.97* | 0.98* | 0.97* | 0.53 | 0.96* | 0.99** | 0.93 |

Notes: *Correlation is significant at the 0.05 level (2-tailed). **Correlation is significant at the 0.01 level (2-tailed).
However, these factors, which are known to increase Rs, are sensitive to shifts in climate conditions, such as rising temperatures\textsuperscript{32,33}. Meanwhile, \( Q_{10} \) is a major source of uncertainty in assessing carbon budgets using carbon cycle models\textsuperscript{99,100}. Therefore, it is important to identify the \( Q_{10} \) of different ecosystems.

In the current study, SOC content is higher within the southeastern boundary of the QTP than in the Haibei alpine meadow located in the inner QTP\textsuperscript{31,38}, and a little higher than in alpine grasslands with an altitude of over 4500 m located in the Tibet\textsuperscript{63–65}, but is lower than in the Zoigê alpine wetland located at the eastern edge of the QTP\textsuperscript{106,107}. The Rs rate in the alpine wetland meadows in this study is roughly similar to that of degraded grassland located at the northeastern edge of the QTP\textsuperscript{108}, but it is higher than in the Haibei alpine meadow\textsuperscript{56,51,88,94} and the inner QTP\textsuperscript{31,57,76,98}.

The \( Q_{10} \) value of the alpine wetland meadow in this study is higher than in the Haibei alpine meadow and other alpine regions (range 1.3–5.6)\textsuperscript{36,51,86,94,109–111}, the Zoigê alpine wetland\textsuperscript{106,107}, the inner QTP (range 1.05–2.81)\textsuperscript{11,75,98}, and the global average (range 1.3–3.3)\textsuperscript{85,112}. Many studies have suggested that \( Q_{10} \) declines with increasing temperature\textsuperscript{31,111–113}. It is worth noting, however, that both the mean temperature and the \( Q_{10} \) are higher in this study area than in the Haibei region mentioned above. Furthermore, \( Q_{10} \) correlates significantly with SOC in our study (Table 3), but the \( Q_{10} \) is higher than in the Zoigê alpine wetland because of the higher SOC content at our study site.

Together, these results suggest that Rs sensitivity to temperature is greater in alpine wetland meadow ecosystems located in the boundary region of the QTP than in other zones. Also, we speculated that \( Q_{10} \) of alpine wetland meadow is probable greater at the boundary region than inner region of the Qinghai-Tibet Plateau, and should be a more sensitive indicator in the studying of climate change in this zone.

**Conclusions**

To the best of our knowledge, this study is the first to observe Rs on diurnal and seasonal time scales, and to quantitatively analyze Rs and \( Q_{10} \) at four different levels of alpine wetland meadow degradation in the Napa Lake region of Shangri-La, at the southeastern edge of the QTP.

In summary, we found that vegetation degradation markedly altered the Rs of the alpine wetland meadow. Rs decreased by more than 50% when degradation intensity increased from NDM to SDM. On the scale of diurnal variation, Rs correlated significantly with Ts at the 0–5 cm soil depth (\( p < 0.05 \)), but not at the seasonal scale. The \( Q_{10} \) value of Rs decreased significantly with an increase in degradation from NDM to SDM during every season. Rs and \( Q_{10} \) were higher during the MGS than during the EGS and LGS at every level of degradation. These results indicate that vegetation condition plays an important role in controlling Rs and \( Q_{10} \).

**Materials and Methods**

**Site description.** This study was performed at the Napa Lake basin in Shangri-La County (N27°49′–27°55′, E99°37′–99°40′; mean altitude 3350 m), which lies at the southeastern edge of the QTP in northwestern Yunnan province, China (Fig. 9). Napa Lake is a typical plateau lake found on the Tungui plateau. It is situated in a grabbing basin in the alpine and gorge region of the Hengduan Mountains.

The study region has a cold and moist subtropical southwestern monsoon climate that is influenced by the region’s high altitude and plateau landscape. Mean annual temperature is 6.4 °C, mean monthly minimum and maximum temperatures are –3.6 °C in January and 13.2 °C in July, mean annual precipitation is 632.4 mm.

The annual range in temperature is small, but the daily range in temperature range is large. The rainy season lasts from June to October and the dry season lasts from November to May. The growing season lasts from about May to September. Soil types in the region are mainly swamp soil, peat soil, and alpine meadow soil.

Vast areas of alpine wetland meadows are distributed around Napa Lake, with dominant plant species including *Blysmus sinocompressus*, *Carex mulliensis*, *Poa szechuensis*, *Pedicularis longiflora*, *Kobresia bellardii*, and *Potentilla anserina*. Villages are located far from the lakeside, while alpine meadows and farmland planted with *Hordeum vulgare* are close to the lakeside. As altitude increases, vegetation succeeds gradually from hard-leaf evergreen and broad-leaved forests, to alpine shrubs, to alpine pine forest, and to spruce and fir forests. Dominant species include *Crataegus oregia*, *Populus rotundfolia*, *Sabina squamata*, *Pinus densata*, *Picea asperata* Mast., and *Abies forrestii*.

**Plot surveys and Rs measurements.** Study plots. We classified alpine wetland meadows within the study area into four levels of degradation based on the presence of fending, grazing activity, tourism disturbance, and aboveground biomass and vegetation cover: non-degraded meadow (NDM), lightly-degraded meadow (LDM), moderately-degraded meadow (MDM), and severely-degraded meadow (SDM) (Table 1).

Each of the four levels of alpine wetland meadow degradation severity contained three study plots (100 m × 100 m). All plots are located adjacent to a lake, but are exposed to water year-round, and do not experience periodic flooding. By correlating the degree of degradation in the plots with vegetation cover, biomass, and species composition, we can determine the effects of grazing and tourism on the meadows.

Vegetation surveys and soil organic carbon (SOC) measurements. To further characterize and verify the degree of degradation in the different plots, we sampled aboveground biomass, vegetation cover, and species composition within one randomly-placed 1 m × 1 m frame on every plot every season. The leaf area index (LAI) within the sampling frames was determined using a plant canopy analyzer (LI-COR LAI-2200 Plant Canopy Analyzer, LI-Cor, Lincoln, Nebraska, USA).

Plant samples from each sampling frame were dried in an oven at 65 °C for at least 48 h and then weighed. Other plant samples from the sampling frames were combined by plot of same degradation level, dried at 105 °C for 15 min, and then dried at 65 °C for 48 h to measure the carbon content of the vegetation.
One soil profile with a depth of 50 cm were collected from each plot during the experimental period, and the SOC content of the soil at different depths (0–10 cm, 10–20 cm, 20–30 cm, 30–40 cm, and 40–50 cm) was analyzed for every degradation level.

Rs measurement. Rs was measured in each plot using an automated CO$_2$ efflux system (Li-8100, LI-COR Inc., Lincoln, NE, USA) in May (early growing season, EGS), July (mid growing season, MGS), and September (late growing season, LGS) over the course of the full growing season (May to September) in 2014 and 2015.

CO$_2$ measurements were collected from each of the four plots for a period of 24 h using a Li-8100 automated soil CO$_2$ flux system with a No. 103 chamber (Li-Cor Inc., Lincoln, NE, USA) to determine diurnal Rs during the growing season in the two years of the study. During the measurements, all chambers were placed on collars with a 20 cm inside diameter and a 10 cm height. The collars had been inserted 5 cm into the soil at least three days prior to measurement. Aboveground vegetation was clipped from the soil surface inside the collars before measuring Rs. All collars were left at the plots during the entire experimental period. Rs include respiration from plant roots and microbes.

Diurnal variations in Rs were recorded automatically every half hour from 7:00 am on the first day to 7:00 am on the next day. The duration of each automatic measurement was about 3 min, which included 15 s dead band, 45 s pre-purge, 45 s post-purge, and 90 s observation. The linear increase in CO$_2$ concentration within the chamber was used to estimate Rs. We simultaneously measured Ts and soil moisture at 5 cm soil depth near each collar using the temperature and moisture sensors of the Li-8100 System.

Statistical analysis. An exponential equation was used to describe the relationship between Rs and Ts:

$$Rs = ae^{bt_s}$$

where, Rs is soil respiration rate ($\mu$mol·m$^{-2}$·s$^{-1}$), Ts is soil temperature (°C) at 5 cm depth, and a and b are fitted parameters.

The sensitivity of Rs to Ts can be defined as the increase in the Rs rate that results from each 10 °C increase in Ts. This sensitivity ($Q_{10}$) can be calculated as follows:

$$Q_{10} = e^{10b}$$

Figure 9. Location of Shangri-La on the Qinghai-Tibet Plateau.
One-way ANOVA was used to compare vegetation condition, SOC, and Rs at different levels of degradation. Exponential regression was used to evaluate the relationship between Rs and Ts in every plot. Linear regression was used to correlate vegetation, SOC, Rs, and Q10. All statistical analyses were performed using SPSS 13.0 software (SPSS for Windows, Chicago, IL, USA). Differences were considered significant when p < 0.05.

Data Availability

The datasets generated during and/or analysed during the current study are available from the corresponding author on reasonable request.

References

1. Bhattacharya, S. S. et al. A review on the role of organic inputs in maintaining the soil carbon pool of the terrestrial ecosystem. J. Environ. Manage. 167, 214–227, https://doi.org/10.1016/j.jenvman.2015.09.042 (2016).

2. Kuske, T. et al. Atmospheric tomography as a tool for quantification of CO2 emissions from potential surface leaks: signal processing workflow for a low accuracy sensor array. Energy Proc. 37, 4065–4076, https://doi.org/10.1016/j.egypro.2013.06.307 (2013).

3. Le Quére, C. et al. Global Carbon Budget 2015. Earth Syst. Sci. Data 7, 349–396, https://doi.org/10.5194/essd-7-349-2015 (2015).

4. Houghton, R. A. Balancing the global carbon budget. Annu. Rev. Earth Pl. Sc. 35, 313–347, https://doi.org/10.1146/annurev.earth.35.031306.140057 (2007).

5. Boden, T. A.,Marland, G. & Andres, R. J. Global, regional, and national fossil-fuel CO2 emissions: Oak Ridge, TN: Carbon Dioxide Information Analysis Center, ORNL, https://doi.org/10.3334/CDIAC/00001_V2010 (2010).

6. IPCC Climate Change: The Assessment Reports of the Intergovernmental Panel on Climate Change, https://doi.org/10.1007/1-4020-2366-8_109 (2013).

7. Bolin, B. Changing global biogeochemistry. In: Brewer, P. G. (Ed.), Oceanography: The Present and the Future Springer, US, pp. 305–326, https://doi.org/10.1007/978-1-4612-5440-9_18 (1983).

8. Houghton, R. A. & Woodwell, G. M. Global climatic change. Sci. Am. 260, 18–26, https://doi.org/10.1038/scientificamerican0489-36 (1989).

9. Schlesinger, W. H. & Andrews, J. A. Soil respiration and the global carbon cycle. Biogeochemistry 48, 7–20, https://doi.org/10.1023/a:1006247623877 (2000).

10. Ryan, M. G. & Law, B. E. Interpreting, measuring, and modeling soil respiration. Biogeochemistry 48, 1–17, https://doi.org/10.1023/a:1006247623877 (2000).

11. Jonathan, G., Martin, P. & Bolstad, V. Variation of soil respiration at three spatial scales: the importance of a model-data integration framework for data interpretation. Plant Nutr. Soil Sc. 171, 344–354, https://doi.org/10.1007/jpln.200700075 (2008).

12. Bond-Lamberty, B. & Thomson, A. A global database of soil respiration data. Biogeosciences 7, 1915–1926, https://doi.org/10.5194/bg-7-1915-2010 (2010).

13. Jonathan, G., Martin, P. & Bolstad, V. Variation of soil respiration at three spatial scales: Components within measurements, intra-site variation and patterns on the landscape. Soil Biol. Biochem. 41, 530–543, https://doi.org/10.1016/j.soilbio.2008.12.012 (2009).

14. Trettin, C. C. & Jorgensen, M. F. Carbon cycling in wetland forest soils. In: Kimble, J. M. Birdsey, R. & Lal, R. (Eds) The Potential of U.S. Forest Soils to Sequester Carbon and Mitigate the Greenhouse Effect. pp. 311–331 (CRC Press, Boca Raton, 2003).

15. Schellbauer, I. L. et al. Seasonal differences in the CO2 exchange of a short-hydroperiod Florida Everglades marsh. Agric. Forest Meteorol. 150, 994–1006, https://doi.org/10.1016/j.agrformet.2010.03.005 (2010).

16. Miao, G. et al. The effects of water table fluctuation on soil respiration in a lower coastal plain forested wetland in the southeastern U.S. J. Geophys. Res. Biogeosci. 118, 1748–1762, https://doi.org/10.1002/jgrg.20254 (2013).

17. Sun, Z. G. et al. China’s wetlands conservation: Achievements in the Eleventh 5-year Plan (2006–2010) and challenges in the Twelfth 5-year Plan (2011–2015). Environ. Manag. J., https://doi.org/10.30638/ermj.2014.043, (2014).

18. Bridges, E. M. World Soils. pp. 1–176, (Cambridge, New York; Melbourne: Cambridge University Press, UK, 1978).

19. Laffuer, P. M., Roulet, N. T. & Admar, S. W. Annual cycle of CO2 exchange at a bog peatland. J. Geophys Res. 106, 3071–3082, https://doi.org/10.1029/2000JD900588 (2001).

20. Muro, J. R. & DaLaune, R. D. Biogeochemistry of Wetlands: Science and Application. CRC Press, Boca Raton, pp. 1–800, https://doi.org/10.2136/usaas2008.0015br (2008).

21. Ciais, P. et al. CO2 fluxes from peat in boreal mires under varying temperature and moisture conditions. J. Ecol. 84, 219–228, https://doi.org/10.2307/2261357 (1996).

22. Szymanski, A. R. & Bayley, S. E. Net aboveground primary production along a bog-rich fen gradient in central Alberta, Canada. Wetlands 16, 467–476, https://doi.org/10.1007/bf03161336 (1996).

23. Thormann, M. N. & Bayley, S. E. Aboveground plant production and nutrient content of the vegetation in six peatlands in Alberta, Canada. Plant Ecol. 131, 1–16, https://doi.org/10.1023/a:1009736005824 (1997).

24. Ström, L. & Christensen, T. R. Below ground carbon turnover and greenhouse gas exchanges in a sub-arctic wetland. Soil Biol. Biochem. 39, 1689–1698, https://doi.org/10.1016/j.soilbio.2007.01.019 (2007).

25. Bernal, B. & Mitsch, W. J. Comparing carbon sequestration in temperate freshwater wetland communities. Glob. Change Biol. 18, 1636–1647, https://doi.org/10.1111/gcb.12486.2011.02619.x (2012).

26. Wang, J. et al. CO2 efflux under different grazing managements on subalpine meadows of Shangri-La, Northwest Yunnan Province, China. Acta Ecol. Sin. 28, 3574–3583, https://doi.org/10.5194/s1872-2032(08)60074-8 (2008).

27. Liu, X. & Chen, B. Climatic warming in the Tibetan Plateau during recent decades. Int. J. Climatol. 20, 1729–1742, 10.1002/1097-0088(20001130)20:14<1729::AID-JOC556>3.0.CO;2-Y (2000).

28. Cai, Q. Q. et al. Vertical distribution of soil organic carbon and carbon storage under different hydrologic conditions in Zoige Alpine Kobresia Meadows wetland. Sci. Silvae Sin. 49, 9–16, https://doi.org/10.11077/j.1001-7488.20130302 (2013).
Acknowledgements
The authors thank Xi Zhang, Xuan Li, Xuchen Ma and Huili Zhao for their work in the field. This work was supported by the National Natural Science Foundation of China (grant no. 31360122), the National Key R&D Program of China (grant no. 2017YFC0506605), Construction and Application Demonstration of Regional Ecological Quality Comprehensive Monitoring Technology System (grant no. 2017YFC0503806).

Author Contributions
G.J. designed the study. L.Z., W.L. and X.D. performed the field data collection. L.Z. wrote the manuscript. Z.C., F.C. and L.D. participated to data analysis and revised this manuscript. All authors discussed results and commented on the manuscript.

Additional Information
Competing Interests: The authors declare no competing interests.

Publisher's note: Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.

Open Access This article is licensed under a Creative Commons Attribution 4.0 International License, which permits use, sharing, adaptation, distribution and reproduction in any medium or format, as long as you give appropriate credit to the original author(s) and the source, provide a link to the Creative Commons license, and indicate if changes were made. The images or other third party material in this article are included in the article’s Creative Commons license, unless indicated otherwise in a credit line to the material. If material is not included in the article’s Creative Commons license and your intended use is not permitted by statutory regulation or exceeds the permitted use, you will need to obtain permission directly from the copyright holder. To view a copy of this license, visit http://creativecommons.org/licenses/by/4.0/.

© The Author(s) 2019