



Eucalypt Leaf Litter and a Drying-Rewetting Cycle Shape Wetland Soil-Water Nutrient Dynamics: A Laboratory Microcosm Study

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Received: 9 February 2026 / Accepted: 5 March 2026 / Published online: 23 March 2026
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Abstract

This study used laboratory column microcosms with River Murray (Australia) wetland soils and water to examine how drying-rewetting (DRW) and river red gum (*Eucalyptus camaldulensis*) litter inputs interactively affect nitrogen (N), phosphorus (P), and carbon dynamics across soil, pore water, and overlying water. We quantified N and P species and dissolved organic carbon (DOC) under two water regimes (constant submergence versus DRW) with and without litter addition to resolve cross-compartment responses at the aquatic-terrestrial interface. DRW enhanced net inorganic-N mobilization, resulting in pronounced post-rewetting pulses in pore-water and overlying-water N (especially NH_4^+) compared to constant submergence, with concentrations gradually converging during prolonged inundation. Conversely, total P and inorganic P (P_i) were higher under constant submergence, consistent with sustained reducing conditions promoting P release in solution. Although litter inputs did not substantially alter total nutrient levels in overlying water, they shifted pore-water chemistry by increasing the proportion of inorganic P and strongly elevating DOC. Following rewetting, DOC increased sharply in litter-amended treatments, ~2.9-fold in overlying water and ~16.8-fold in pore water, compared with no-litter treatments. Litter addition also reduced nitrate availability in soil and pore water during resubmergence, consistent with stronger microbial N retention and/or NO_3^- consumption under rapidly re-established reducing microsites after rewetting. Overall, DRW and litter inputs interact to amplify short-lived C and N pulses while modifying P speciation, with implications for predicting nutrient availability and water-quality risk in wetlands experiencing increasingly variable hydrological regimes.

Keywords Wetlands · Nutrient cycling · Dry-rewetting cycles · River red gum litter · Aquatic-terrestrial interface · Microbial activity · Nitrogen mobilization

Introduction

Mechanistically, DRW can strongly influence nutrients and C fluxes, especially when plant litter is present. During drying, water limitation constrains diffusion and overall microbial respiration, yet partial litter decomposition and the accumulation of microbial biomass and metabolites on litter surfaces can still occur (Xu et al. 2016; Zhang et al. 2024). Upon rewetting, physical fragmentation, microbial turnover, and enhanced enzyme activity collectively drive a pulse of

dissolved organic matter and inorganic nutrients from soil and litter into both pore water and the overlying water column (Gómez et al. 2012; Rodríguez et al. 2019; Marschner 2021). In addition, drying increases oxygen penetration into surface soil, whereas rewetting and renewed submergence can rapidly deplete oxygen and re-establish anoxic conditions, creating redox oscillations. These aerobic-anaerobic transitions may promote nitrification-denitrification and ammonification, affecting whether inorganic N is produced or removed (White and Reddy 2009; Matthews et al. 2023). P may similarly undergo adsorption-desorption or precipitation, influenced by redox fluctuations (Schönbrunner et al. 2012; Gu et al. 2018).

Furthermore, water-level regulation and hydrological variations in certain wetlands, such as constructed or reservoir-type systems, can amplify these DRW effects on C and nutrient cycling. DRW reshape sediment redox conditions,

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organic matter accumulation, and microbial community structures (Li et al. 2024), and DRW events within fluctuation zones can simultaneously enhance C mineralization and greenhouse gas emissions (Petrakis et al. 2017). However, it remains poorly understood how DRW interacts with plant litter inputs to jointly regulate C, N and P dynamics across multiple compartments (soil, pore water, and overlying water). This knowledge gap constrains our ability to predict nutrient availability and water-quality risk under increasingly dynamic hydrological regimes (de Vicente 2021). In particular, field observations often lack the temporal resolution and minimally invasive access required to resolve inter-compartment coupling and disentangle litter-driven responses from sediment processes (Kinsman-Costello et al. 2016; Leitner et al. 2017). Meanwhile, integrated laboratory microcosm experiments that quantify soil, pore water, and overlying water simultaneously remain limited. Such a holistic framework would provide deeper insights into how DRW influences the cycling and availability of C, N, and P across compartments both with and without organic litter (Xiang et al. 2008; Luo et al. 2020; Chen et al. 2020).

The River Murray is Australia's largest river system, with a catchment area of over 1 million km². River Murray wetlands typically maintain hydrological connectivity with the River Murray when river flows exceed approximately 5×10^6 m³ day⁻¹, corresponding to water levels between +0.5 and +1.0 m AHD (Australian Height Datum) in the lower reaches of the system. Below these flow thresholds, particularly during drought conditions, the wetland becomes disconnected, leading to substantial drying and biogeochemical shifts (Paul et al. 2011). In the Murray-Darling Basin of Australia, seasonal droughts can persist for several weeks to months, followed by flood pulses that temporarily inundate wetlands (CSIRO 2008; Finlayson et al. 2013). These DRW events create dynamic redox transitions, which can fundamentally alter N and P forms and fluxes in wetland soil and water. Furthermore, the leaf litter of river red gum (*Eucalyptus camaldulensis*), a dominant riparian species on the River Murray, typically has a high carbon-to-nitrogen (C: N) ratio (Esslemont et al. 2007). Such C-rich litter can increase microbial N demand and promote net immobilisation of inorganic N, particularly when readily degradable C is available (Sawada et al. 2015). However, the net response depends on litter biochemical quality: recalcitrant compounds (e.g., lignin) and phenolic or tannin constituents in eucalypt litter or leachates may slow decomposition and suppress microbial activity, thereby attenuating or delaying immobilisation and mineralisation (Vigo et al. 2024; Li et al. 2025). Concurrently, DRW-induced shifts toward anoxic conditions may facilitate microbial processes, such as iron reduction, releasing phosphate previously bound in sediments (McPhan et al. 2022). Nevertheless, there is a

knowledge gap regarding how high C: N ratio litter inputs such as the river red gum leaves in Australian wetlands can simultaneously affect nutrient cycling across soil, pore water, and overlying water under DRW conditions.

The aim of this study was to conduct an integrated laboratory microcosm experiment to assess nutrient forms and concentrations in wetland soils, pore water and overlying water under different water conditions (constant submergence and DRW) with and without plant litter. We hypothesized that (i) compared to constant submergence, DRW would increase net mobilization and alter speciation/partitioning of N, P and C from soil to pore and overlying water, and (ii) the addition of litter would enhance these concurrent changes in concentrations and speciation across soil, pore water, and overlying water, particularly under DRW conditions.

Materials and Methods

Experimental Setup

Soil and water for the experiment were collected at the Teal Flat wetland on the River Murray situated about 36 km north of Murray Bridge, South Australia (34°53'07'S, 139°33'33'E). The local climate is semi-arid with periodic drought (CSIRO 2008). Mean annual maximum and minimum temperatures are 23.2 °C and 9.0 °C, respectively. The mean annual precipitation is 306 mm (Australian Bureau of Meteorology). Surface soil (0–15 cm depth) was collected in November 2022 from the open water area (~1 m water depth, 14 m from the wetland edge), air-dried, sieved (<2 mm), and homogenized. Wetland water was taken from the same site and stored at 4 °C until use. River red gum leaf litter, a dominant riparian species on the River Murray (McPhan et al. 2022), was collected from the wetland margin at Teal Flat. At collection, litter was not submerged and leaves were dry to the touch or only slightly damp (e.g., minor surface moisture likely from dew), with no standing water on leaf surfaces. Because the residence time of leaves on the soil surface could not be determined in the field, we standardised litter quality by selecting recently senesced leaves that were structurally intact and visually undecomposed (no obvious fragmentation, mould growth, or advanced darkening or softening). Selected leaves were pooled and homogenised to create a composite litter batch. Leaves were oven-dried at 40 °C to constant mass and stored dry until use. Prior to addition, leaves were cut with clean scissors into 1–2 cm pieces to facilitate homogeneous amendment within the columns and to standardise particle size across replicates. Chemical properties of the river red gum litter (e.g., total C, total N, total P and stoichiometric

ratios) are summarised in Table 1. The soil had a sandy texture (87% sand, 9% clay, and 4% silt) with a bulk density of $1.67 \pm 0.02 \text{ g cm}^{-3}$ (Table 1). The wetland water itself was characterized by pH of 6.67 ± 0.01 , dissolved organic carbon (DOC) concentration of $17.2 \pm 1.1 \text{ mg L}^{-1}$, total nitrogen (TN) content of $0.25 \pm 0.02 \text{ mg L}^{-1}$ and total phosphorus (TP) concentration of $0.14 \pm 0.01 \text{ mg L}^{-1}$ (Table S1). The C: N ratio of the litter was 45.

This experiment comprised two litter treatments (with-out litter addition and river red gum litter addition) and two water regimes (constant submergence and DRW), arranged in a randomised block design with four replicates (Fig. 1). A total of 24 soil-water microcosms were established. Each consisted of an uncapped PVC tube (25 cm high, 4.4 cm inner diameter) filled with approximately 381 g of air-dried soil to a depth of 15 cm. Wetland water was carefully added along the tube walls until reaching a depth of 5 cm above the soil surface ($\sim 76 \text{ mL}$ overlying water). In addition, after the overlying water was removed (decanted to the soil surface), the water retained within the packed soil was estimated

gravimetrically, yielding an operational pore-water volume of approximately 100 mL per tube. For litter-addition columns, 1.8 g of dried river red gum litter ($\sim 12 \text{ t ha}^{-1}$ equivalent) was applied evenly on the soil surface (Gibbs et al. 2022).

All columns were incubated at 20°C in the dark for eight weeks, and progressed through three distinct phases (Fig. 1a): Phase 1, Days 1–14 (First submergence): All 24 columns were uniformly submerged to 5 cm above the soil surface to establish consistent starting conditions. Phase 2, Days 15–42 (Drying period): “Drying” refers to removal of the overlying water and subsequent evaporative moisture loss from intact columns under laboratory conditions, rather than complete desiccation. DRW columns had the overlying water removed and were allowed to dry naturally over four weeks. In contrast, constant submergence columns were maintained at the original 5 cm water depth, with water replenished weekly to compensate for evaporation. Phase 3, Days 43–56 (Resubmergence): Overlying-water depth was standardised to 5 cm in all columns for two weeks, by

Table 1 Properties of topsoil (0–15 cm) in the Teal Flat Wetland, and river red gum litter. Mean $\pm$ SE ($n=3$)

	pH (1:5 H_2O)	Total N (mg g^{-1})	Total C (mg g^{-1})	Total P (mg g^{-1})	Na (mg kg^{-1})	K (mg kg^{-1})	Ca (mg kg^{-1})	Al (mg kg^{-1})	Fe (mg kg^{-1})
Wetland Soil	7.71 ± 0.01	0.31 ± 0.1	3.9 ± 0.4	0.24 ± 0.2	0.4 ± 0.1	0.7 ± 0.1	1.4 ± 0.2	3.8 ± 0.4	3.6 ± 0.3
River red gum litter	-	10.8 ± 0.2	487.2 ± 31.1	0.60 ± 0.1	1.9 ± 0.1	1.7 ± 0.2	23.1 ± 1.2	<0.003	0.1 ± 0.0

(A) Laboratory Microcosm Design and Sampling Schedule

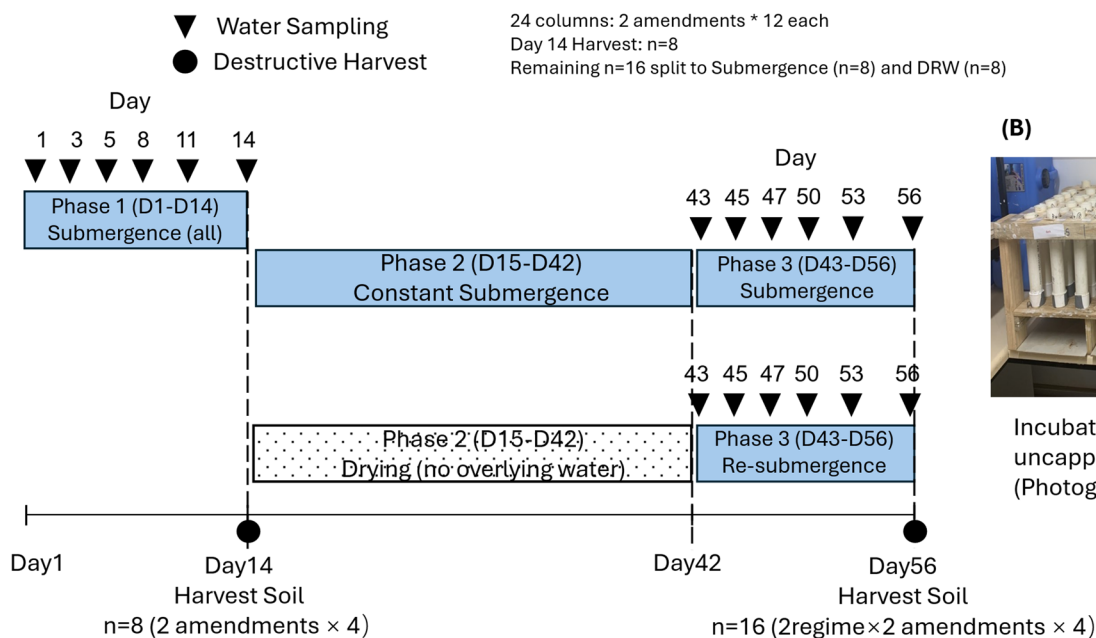
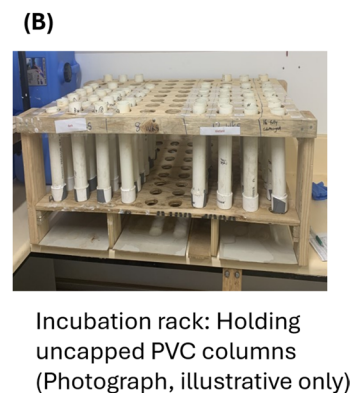


Fig. 1 Laboratory soil-water microcosm experimental design and sampling schedule. (A) Bars denote the water regime applied over time (submergence vs. drying in the DRW treatment), with water-sampling days (triangles) and destructive harvests (circles). A cross-section icon

is provided to indicate the column components (soil, overlying water, and the litter layer in litter-addition treatments). (B) Incubation rack holding uncapped PVC columns (photograph; illustrative only)



re-submerging DRW columns and maintaining constant-submergence columns at 5 cm.

Overlying water and pore water samples (~10 mL each) were collected for chemical analysis on Days 1, 3, 5, 8, 11, and 14 (Phase 1), and Days 43, 45, 47, 50, 53, and 56 (Phase 3). After each sampling event or weekly during Phase 2 (for constantly submerged columns), additional wetland water was added to maintain consistent water depth (Fig. 1). Pore water was collected with Rhizon samplers (Rhizon CSS 19.21.23 F; Rhizosphere Research Products, Wageningen, Netherlands) inserted vertically to 10 cm depth. These samplers have a PVC/PE tube (12 cm long) and connected to a porous section (pore size of 0.15 μm and length 10 cm). Overlying water was drawn via a syringe attached to a PVC/PE tube, then filtered through 0.45 μm membrane filters. All water samples were stored at 4 °C and analysed within 7 days. Soil samples were collected at Days 14 and 56. On Day 14 (end of Phase 1, prior to the divergence of water-regime treatments), eight columns (two litter treatments $\times$ four replicates) were destructively sampled. On Day 56 (end of Phase 3), the remaining sixteen columns (two water regimes $\times$ two litter treatments $\times$ four replicates) were destructively sampled. At each sampling time, the soil columns were carefully removed from the PVC tubes, and any litter on the soil surface was gently separated and discarded to avoid contamination. The soil was then homogenized and immediately placed into sterile plastic bags, sealed and stored at -20 °C until further analysis.

Chemical Analyses

For soil analyses, soil pH was measured in a 1:5 soil: water (w/v) extract (George and David 2010). Soil redox potential (Eh) was measured by inserting the probe (Model IJ64, Ionode, Australia) into the soil without removing overlying water. Eh was recorded at the key sampling days (Days 1, 14, 41 and 56) and reported in mV. For interpretative context, Eh values were used as an operational indicator of redox status: Eh $\geq$ ~+400 mV was considered more oxidising (oxygen may be present), whereas oxygen is expected to be absent at Eh values beginning at or below ~+350 mV; values between +350 and +400 mV were treated as transitional low-oxygen conditions (Pezeshki and DeLaune 2012). Soil maximum water holding capacity was measured using a sintered glass funnel connected to a 1-m water column (Wilke 2005). Soil texture was determined by the hydrometer method (Gee and Or 2002). TN in soil and plant litter was extracted and determined following the Kjeldahl method (McKenzie and Wallace 1954). To determine TP, soil was digested with nitric acid-perchloric acid at a 4:1 ratio and P in the digest was determined by the phosphorvanado-molybdate method

(Hanson 1950). Total organic C (OC) in soil was determined by $\text{K}_2\text{Cr}_2\text{O}_7$ and H_2SO_4 oxidation by titration with acidified $(\text{NH}_4)_2\text{Fe}(\text{SO}_4)_2 \cdot 6\text{H}_2\text{O}$ (Walkley and Black 1934). Soil ammonium (NH_4^+) and nitrate (NO_3^-) were extracted with 2 M KCl solution in a 1:5 soil extractant ratio by shaking for 1 h and were determined as described by Willis et al. (1996) and Miranda et al. (2001), respectively. Available N is the sum of NH_4^+ and NO_3^- . Soil available phosphate was extracted with 0.5 M NaHCO_3 (pH 8.5) at a 1:20 (w: v) ratio by shaking for 16 h at room temperature. The phosphate concentration was determined by the malachite green method (Ohno and Zibilske 1991).

For water analyses, TN and TP were determined by persulfate digestion, followed by spectrophotometric measurement (Armstrong 1963; Ebina et al. 1983). NH_4^+ was measured spectrophotometrically as described for soil samples, while NO_3^- was measured after alumina treatment to remove organic interferences (Cawse 1967). Dissolved organic nitrogen (DON) and phosphorus (DOP) were calculated by subtracting inorganic forms (NH_4^+ , NO_3^- for N; Inorganic P for P) from the total concentrations. DOC was estimated using UV absorbance at 254 nm, a reliable proxy for organic matter concentrations in freshwater (Edzwald et al. 1985). Although UV measurement does not provide the composition of organic compounds, it allows for the sensitive and reliable measurement of organic matter absorbance (Karanfil et al. 2003). Moreover, strong correlations between DOC and UV measurements have been consistently observed in freshwater samples (Edzwald et al. 1985; Gu et al. 1996).

Statistical Analyses

The data was analysed in R 4.4.1 (R Core Team 2024). The residuals (the differences between observed and predicted values) were checked for normality using the Shapiro-Wilk test. Homogeneity of variances among groups was evaluated using Levene's test and by inspection of residuals versus fitted values. TN, available N, DON, TP, Inorganic P (P_i), DOP, DOC and pH in the Phase 1 were analysed by repeated measures one-way ANOVA with a between-subject factor (treatment), and within-subject factor (time). The data of the above parameters between Days 45 and 56 was analysed by repeated measures two-way ANOVA with between-subject factors (litter and water treatments), and a within-subject factor (time). Post hoc pairwise comparisons were performed using Tukey's HSD to identify differences among treatments and sampling times for the factors $\times$ time interaction ($P=0.05$). Redundancy Analysis (RDA) were applied to examine relationships among litter addition vs. no-litter, different water regimes, and the aforementioned

parameters. For Phase 3 (Days 43–56), we fitted a partial RDA conditioning on sampling day (day as a covariate) to remove time-related variation and focus on treatment effects. Significance of the constrained model and terms was assessed using permutation tests.

Results

Soil Pore Water and Overlying Water General Relationships

Redundancy analysis revealed that both water regime and litter addition were the dominant drivers of multivariate nutrient variation in both pore water and overlying water during the final resubmergence phase (Phase 3, Fig. 2 and Table S3). For pore water, the first two axes (RDA1 and RDA2) accounted for 44.3% and 7.6% of the total variation, respectively and the overall constrained fraction was 54.97% (adjusted 56.77%) (Fig. 2a). Among explanatory variables, water regime ($P < 0.001$) and litter addition ($P < 0.001$) exerted the strongest effects on pore water nutrient dynamics. For the overlying water, the first two RDA axes explained 42.5% and 7.4% of the total variance in the log-transformed nutrient data (Fig. 2b). Similarly,

litter addition ($P < 0.001$) and water regime ($P < 0.001$) were the strongest predictors of nutrient dynamics in overlying water. Additionally, a significant interaction between water regime and litter addition ($P < 0.001$) was observed, indicating that the influence of litter input varied with water regime conditions.

Nitrogen

Overall, TN concentrations were lower in overlying water than in pore water regardless of litter and DRW. There were modest differences in TN between the litter treatment and the control without litter addition in both pore and overlying water over time. At Phase 3 (Days 43–56), TN in pore water increased significantly in the litter treatment compared with the control (Fig. 3). Moreover, DRW was associated with a significant ($P < 0.05$) increase in pore-water TN immediately after rewetting and a concomitant decrease ($P < 0.05$) in overlying water, highlighting the transient nature of DRW-induced effects.

During the first submergence (Days 1–14), TN in pore water initially increased until Day 5 in the litter treatment, then decreased thereafter, while the control without litter addition consistently declined from Day 5 onward (Fig. 3a). In overlying water, TN decreased from Day 5 in

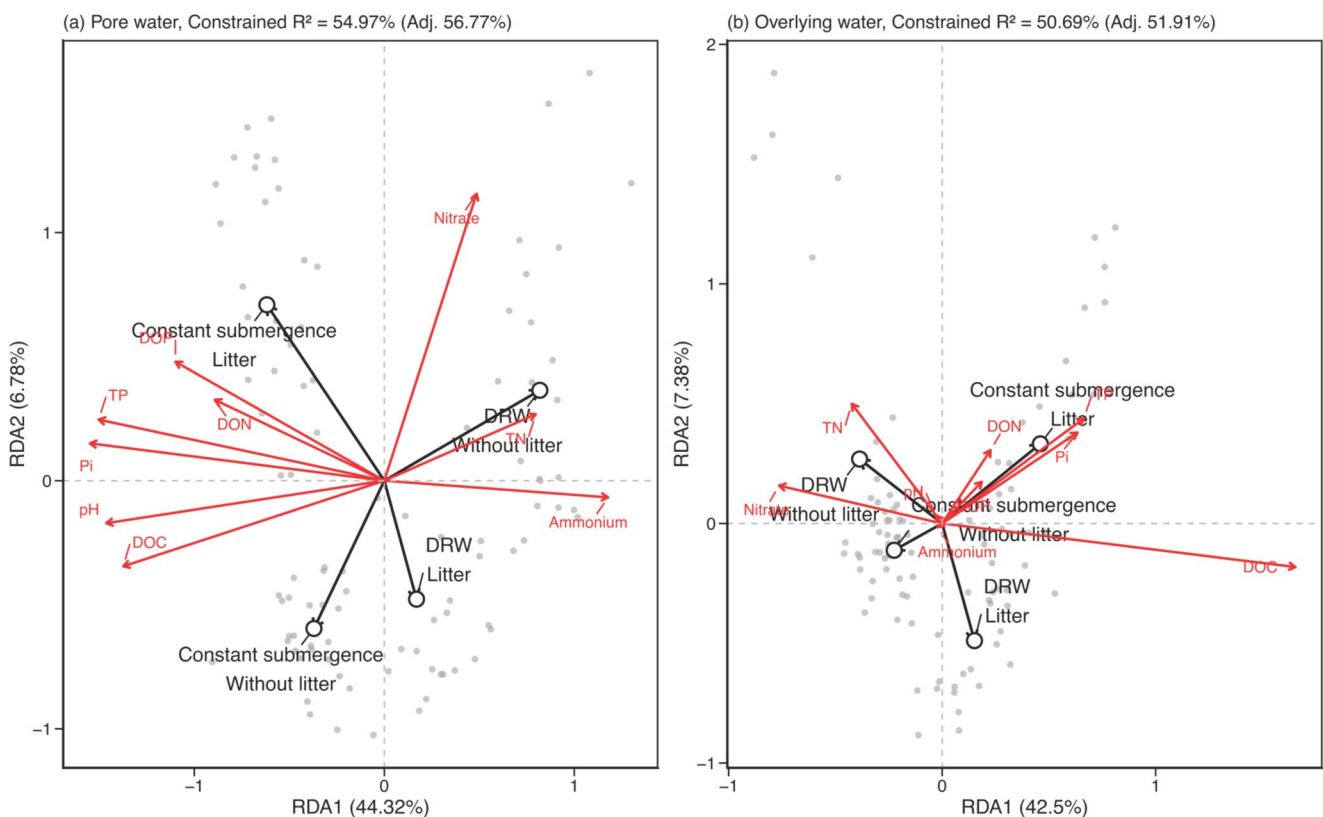


Fig. 2 Biplot of a partial redundancy analysis (RDA) of pore-water (a) and overlying-water (b) nutrient and physicochemical composition during Phase 3 (Days 43–56)

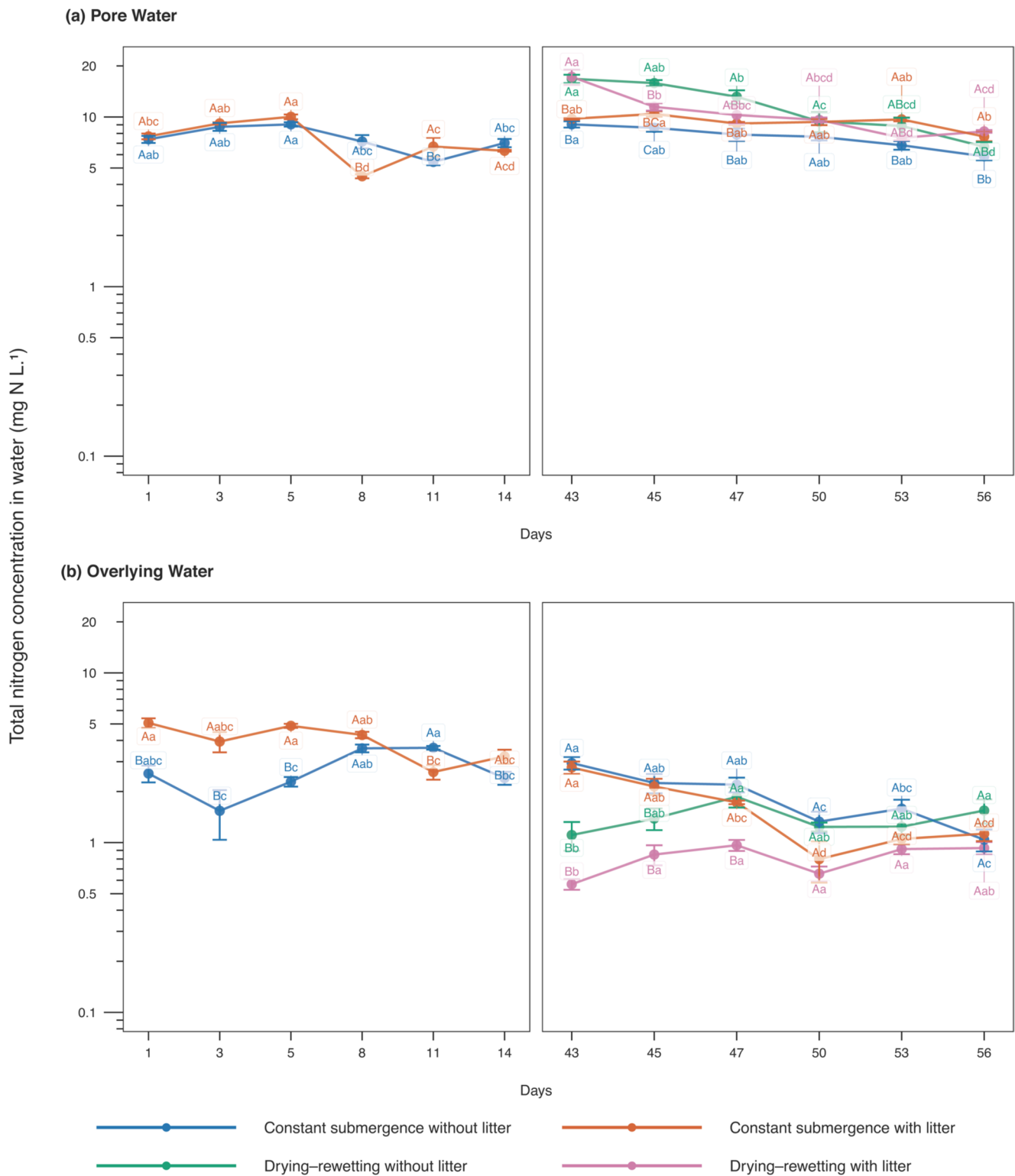


Fig. 3 Total Nitrogen (TN) concentration in pore water **(a)** and overlying water **(b)** during Phase 1 and Phase 3. Different uppercase letters indicate significant differences among treatments at a given sampling time. Different lowercase letters indicate significant differences over

time in a given treatment ($p \leq 0.05$, $n=4$, mean $\pm$ SE). The y-axis is plotted on a log10 scale to accommodate the wide range of values. Symmetric mean $\pm$ SE intervals on the original scale may appear visually asymmetric on the plot

the litter treatment, whereas the control without litter addition increased until Day 11 before decreasing (Fig. 3b). Despite these short-term fluctuations, most sampling days exhibited only minor differences in TN between the litter treatment and control without litter addition during the first submerged period. From Days 43 to 56, under constant submergence, TN concentrations in pore water decreased by about 25–35% for both litter treatment and control without litter addition. After rewetting (Days 43–56) in the DRW treatments, TN in pore water increased ~1.8-times higher than the constant submergence treatments on Day 43, whilst TN in overlying water underwent a sharp decline and the difference between compartments narrowed over time.

Overall, DRW led to a transient surge in the concentrations of NH_4^+ in both pore and overlying water compared with constant submergence regardless of litter addition. DRW increased NH_4^+ by fourfold more than in the constant submergence treatment on Day 43. However, NH_4^+ concentrations subsequently decreased, remaining about twice as high as the constant submergence treatments by Day 56. In contrast under constant submergence, NH_4^+ concentrations in pore water with the litter treatment were 30–60% lower than in the control without litter addition between Day 5 and Day 14 (Fig. S1a).

NH_4^+ concentrations in overlying water were about 10- to 50-fold lower compared to pore water (Fig. S1b). During the first five days of submerged period, NH_4^+ in the control without litter addition was approximately 2 to 7-fold higher than those in the litter treatment. However, no significant differences were observed from Days 8 to 14. Between Day 43 and 56, NH_4^+ concentrations in overlying water remained consistently low under constant submergence conditions, irrespective of litter addition. By contrast, DRW increased NH_4^+ by approximately 9- to 19-fold immediately after rewetting (Day 43) compared with corresponding treatments under constant submergence.

Overall, pore-water NO_3^- concentrations were higher in the litter-addition treatment than in the control without litter during both the first submergence (Days 1–14) and the resubmergence period (Days 43–56), with differences reaching up to ~15× on some sampling days (Fig. S2a; $P < 0.05$). After rewetting on Day 43, a ~7-fold increase NO_3^- concentrations in pore water occurred in the control without litter addition compared to the same measurement on Day 14. However, NO_3^- subsequently decreased in the control without litter addition and converged with the litter treatment by around Day 50. In the overlying water, NO_3^- concentrations were generally similar to pore water levels. However, NO_3^- was generally higher in the control without litter addition, especially under the resubmergence where it was ~3- to 8-fold higher than the litter treatment. Following resubmergence, NO_3^- in the overlying water declined and

became relatively low and similar across treatments from Day 43 to 56 (Fig. S2b).

Despite transient differences, DON concentration in pore water was not significantly impacted by litter addition over the short term (i.e., 14 days). The overall patterns mirrored those observed for NH_4^+ . DON in the control without litter remained stable from Day 45 to Day 50, then sharply increased (~80%) on Day 53, followed by a subsequent decline by Day 56 (Fig. S3a). After rewetting from Day 43, DRW decreased DON by ~95% for the litter treatment and the control without litter between Days 43 and 53, followed by a significant increase on Day 56.

In the overlying water, DON concentrations remained low in the control without litter during the first submergence (Days 1–14) and resubmergence (Days 43–56) (Fig. S3b). Under constant submergence conditions (Days 43–56), DON concentrations in the litter treatment decreased by around 50% from Days 43 to 50, stabilizing thereafter. Except for Day 50, DON concentrations with litter were about two-fold higher than in the control without litter. Following DRW-induced resubmergence (Days 43–56), DON concentrations in the overlying water were consistently low across both litter and control treatments.

Compared with constant submergence conditions, DRW markedly increased the proportion of NH_4^+ within porewater, a trend that persisted through the end of the experiment. This effect was particularly pronounced under litter treatment. In contrast, under constant submergence, the relative fraction of DON in porewater rose gradually over time (Fig. 4a). For overlying water, the proportion of DON increased significantly under litter addition, compared to the control without litter. However, at the immediate onset of rewetting following a dry period, the DON fraction in DRW treatments was not significantly different from that observed under either constant submergence treatment (with or without litter). Notably, in the first day of rewetting, both DRW treatments (with and without litter) exhibited a higher proportion of NH_4^+ in overlying water, which subsequently declined toward levels observed under constant submergence (Fig. 4b). Across all treatments, NH_4^+ consistently remained higher in porewater than in the overlying water, whereas NO_3^- concentrations in porewater, except under litter-addition continuous flooding, were significantly lower relative to NO_3^- in the overlying water.

Phosphorus

Overall, the highest concentration of TP in both pore and overlying water was observed under constant submergence with litter addition, whereas the control without litter had relatively low TP levels, regardless of water regimes

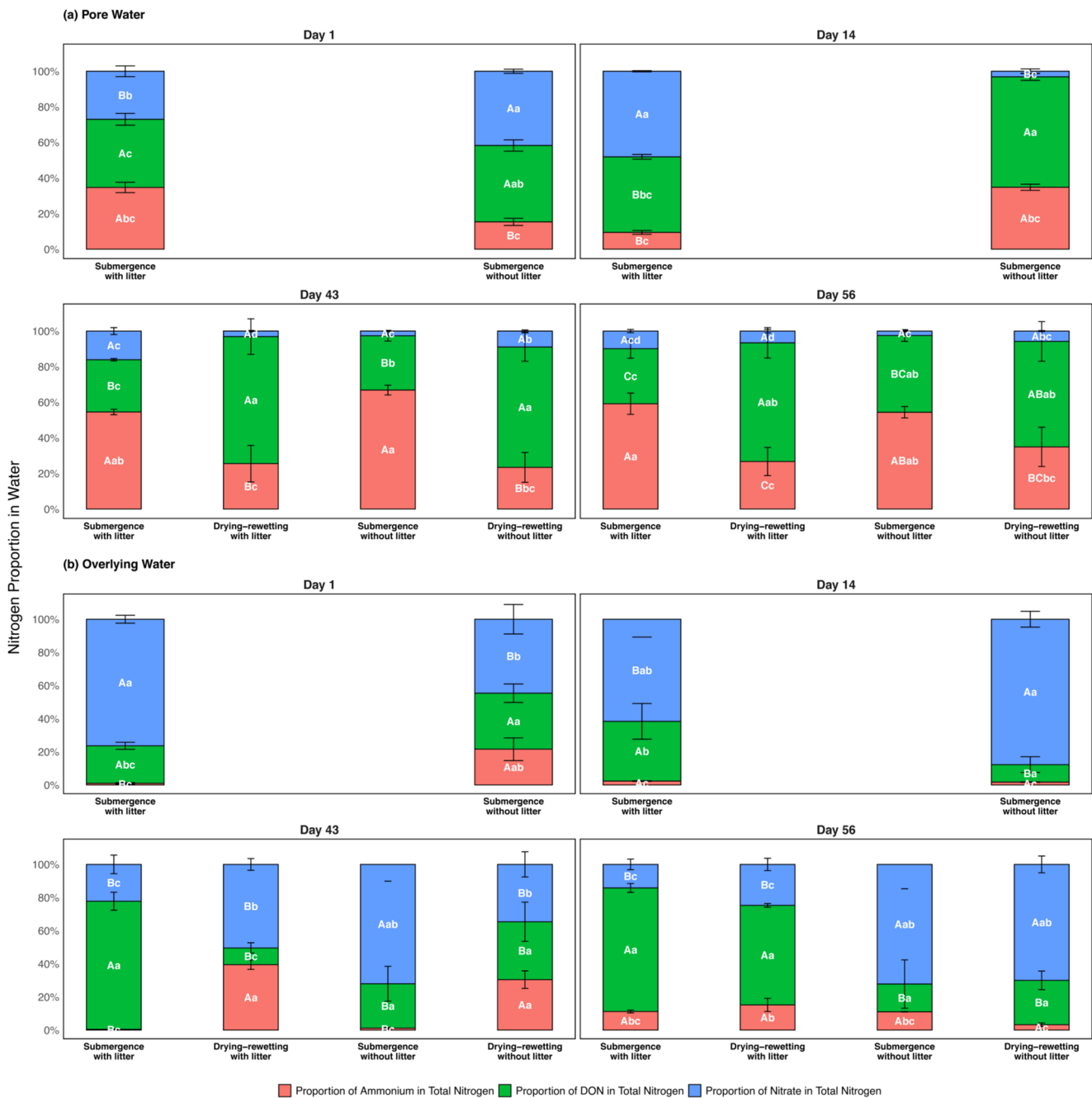


Fig. 4 Percentage of DON, NH_4^+ and NO_3^- in TN in pore water (a) and overlying water (b) during Phase 1 and Phase 3. Different uppercase letters indicate significant differences for a given indicator (DON, NH_4^+ and NO_3^-) among treatments at a given sampling time in the same location (pore water and overlying water); Different lowercase

letters indicate significant differences within the same indicator across different sampling times for the “without litter” treatments (including submergence and drying-rewetting conditions) and the “with litter” treatments (including submergence and drying-rewetting conditions)

(Fig. 5). Meanwhile, the DRW with litter treatment exhibited a clear increase in TP concentrations in pore water over time (Fig. 5a), gradually approaching the levels observed under constant submergence with litter addition. In contrast, the DRW without litter treatment TP level fluctuated but was lower than the litter treatment (Fig. 5b).

During Days 1–14, TP concentrations in pore water remained relatively stable, with only minor differences between litter treatment and the control without litter (Fig. 5a). In contrast, TP concentration in overlying water with litter addition was substantially higher, ranging from about 3- to 30- fold, compared to the control without litter

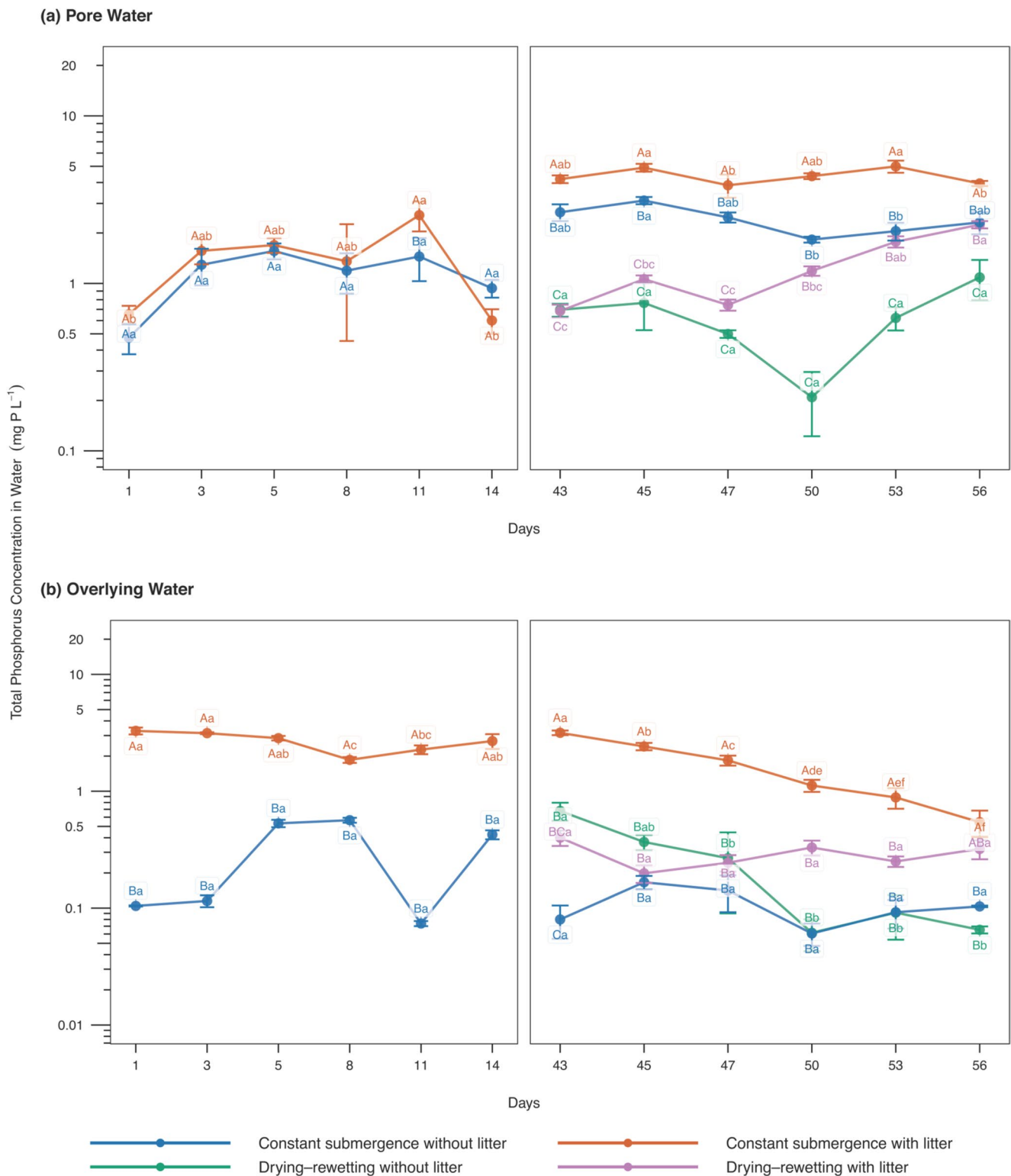


Fig. 5 Total Phosphorus (TP) concentration in pore water (**a**) and overlying water (**b**) during Phase 1 and Phase 3. Different uppercase letters indicate significant differences among treatments at a given sampling time. Different lowercase letters indicate significant differences over

time in a given treatment ($p \leq 0.05$, $n=4$, mean $\pm$ SE). The y-axis is plotted on a logarithmic scale to accommodate the wide range of values. Symmetric mean $\pm$ SE intervals on the original scale may appear visually asymmetric on the plot

(Fig. 5b). Although TP concentrations in overlying water with litter addition decreased by about 45% from Days 1–8, it remained significantly elevated relative to the control on most sampling days.

Under the constant submergence (Days 43–56), TP concentrations in pore water with litter were typically 1.5- to 2.5-fold higher than the control without litter, showing minor temporal fluctuations. Similarly, TP concentrations in overlying water with litter typically remained significantly higher than the control without litter throughout Days 43–56, despite decreasing by up to 80%. Thus, even during the resubmergence, the presence of litter continued to promote higher TP levels than the control without litter. After resubmergence, pore water TP in both treatments initially increased. However, from Day 47 to 56, the litter treatment's pore water TP rose by approximately threefold, while the control without litter increased by about twofold. Consequently, litter-addition TP was 2- to 5.5-fold higher than the control without litter by Day 50.

P_i concentrations in both the pore water and overlying water exhibited temporal trends similar to those observed for TP (Fig. 6 and Fig.S4). However, the control without litter was consistently lower in P_i than the litter-addition treatment ($P < 0.05$), particularly in the pore water where differences increased over time. In the overlying water, P_i generally accounted for $\leq 70\%$ of TP, and while the litter-addition treatment often surpassed the group without litter, some differences were not significant. DRW did not have a significant effect on overlying-water P_i .

Under constant submergence, DOP concentration in pore water accounted for approximately 60–90% of TP in the litter-addition treatment, whereas the control without litter maintained comparatively lower levels. During the first submergence (Days 1–14), pore water DOP in both litter treatment and the control without litter rose to around 3-fold higher than that on Day 1 by Day 3, before reducing (Fig. S5a). Over the constant submergence period, DOP concentration in pore water was generally 1.5–3 times higher in litter-addition treatments compared to the control without litter, except on Days 47 and 56. After resubmergence, pore-water DOP for the control without litter increased from Day 50, reaching $\sim 70\%$ on Day 56 compared to Day 43, and ultimately surpassing the litter treatment by about fourfold on Day 56. Thus, although pore water DOP initially converged across treatments during the early submergence period, it subsequently rose markedly in the unamended soil after rewetting.

In overlying water DOP showed a markedly stronger response to litter addition than to other factors. During the first submergence (Days 1–14), litter-addition treatments showed DOP concentrations approximately 5–90 times greater than the control without litter, peaking on Day 14

at around 1.5-fold higher compared to Day 1 (Fig.S5b). Under constant submerged conditions (Days 43–56), the litter treatment sustained significantly higher DOP (~ 3 – 18 times greater) than the control without litter, with DOP accounting for $\sim 30\%$ – 60% of TP versus only $\sim 10\%$ – 30% without litter. After resubmergence period, DOP for the control without litter concentrations dropped by $\sim 90\%$ from Days 43–56, whereas DOP with litter fluctuated and peaked on Day 47. Consequently, DOP concentrations in overlying water were substantially elevated with litter during short-term submergence and generally remained higher than unamended levels under constant submergence, highlighting the ongoing impact of organic litter on DOP in the water column.

Following litter addition, subsequent DRW lead to an increased proportion of P_i relative to TP in the pore water. In contrast, without litter, DRW results in an increased proportion of DOP in the pore water. Regarding the overlying water, initial submergence is characterized by a relatively low proportion of P_i ($P < 0.05$) (Fig. 6). However, as the wetting period is extended, the differences in P_i proportions between the DRW treatments and constant submergence treatments diminish.

Dissolved Organic Carbon

DOC concentrations were substantially higher under litter treatment and tended to rebound strongly after rewetting, whereas concentrations in the control without litter generally remained low in both pore water and overlying water. Although drying temporarily reduced DOC in both litter treatment and the control without litter, the post-rewetting surge was much more pronounced when litter was present, highlighting the capacity of organic inputs to sustain elevated DOC levels even after moisture fluctuations.

During the first submergence (Days 1–14), pore-water DOC in the litter treatment was approximately 2–3-fold higher than in the control without litter (Fig. 7a). Under constant submergence (Fig. 7b), litter treatment also maintained a similar 2–3-fold increase. Although drying lowered DOC in both treatments, rewetting from Day 43 to 47 caused DOC in the litter treatment to peak, reaching ~ 76 -fold above the control without litter, before stabilizing. Meanwhile, DOC in the DRW without litter treatment remained consistently low (80–95% lower) compared to constant submergence. In the overlying water, the litter-addition DOC similarly remained 11–33 times higher than the control without litter during early submergence, despite a $\sim 75\%$ drop from its initial peak. Under constant submergence, litter-addition DOC gradually declined but was still ~ 2 – 5 times higher

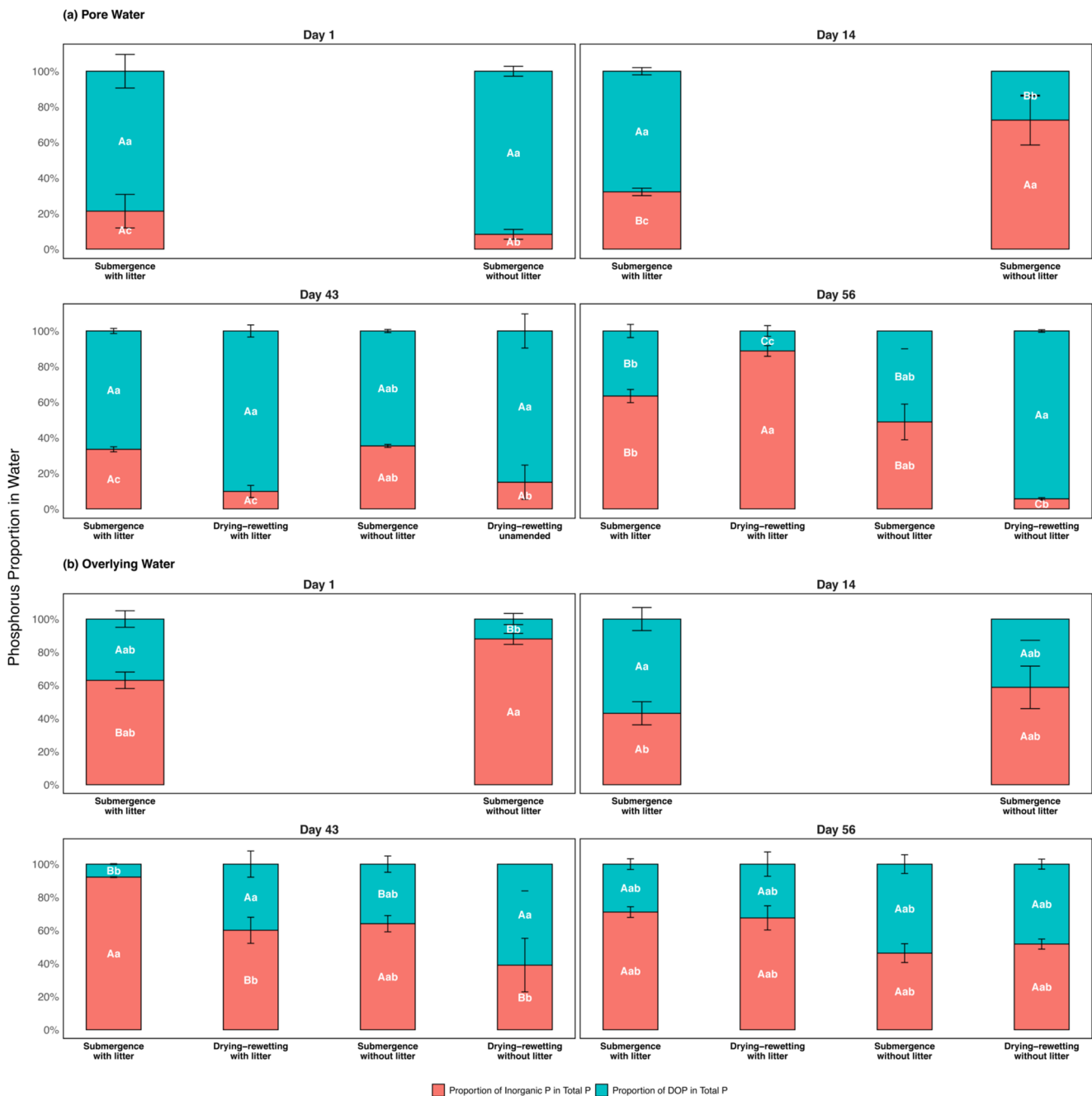


Fig. 6 Percentage of DOP and inorganic P in TP in pore water (a) and overlying water (b) during Phase 1 and Phase 3. Different uppercase letters indicate significant differences for a given indicator (DOP and inorganic P) among treatments at a given sampling time in the same location (pore water and overlying water); Different lowercase letters

indicate significant differences within the same indicator across different sampling times for the “without litter” treatments (including submergence and drying-rewetting conditions) and the “with litter” treatments (including submergence and drying-rewetting conditions)

than unamended, and after rewetting, rose again to ~2–3 times unamended levels.

Water pH and Soil Properties

During the first submergence, pH of pore water remained stable regardless of litter addition. By contrast, after

rewetting, the pH dropped by about two units within the first two days and then stabilized around 5.5 until the end of re-submergence. Overlying water pH initially showed larger differences with litter addition, although it tended to converge or remain only ~0.4–1.5 units below the control without litter over time. In the first submergence (Days 1–14), the pH in the litter treatment gradually increased whereas

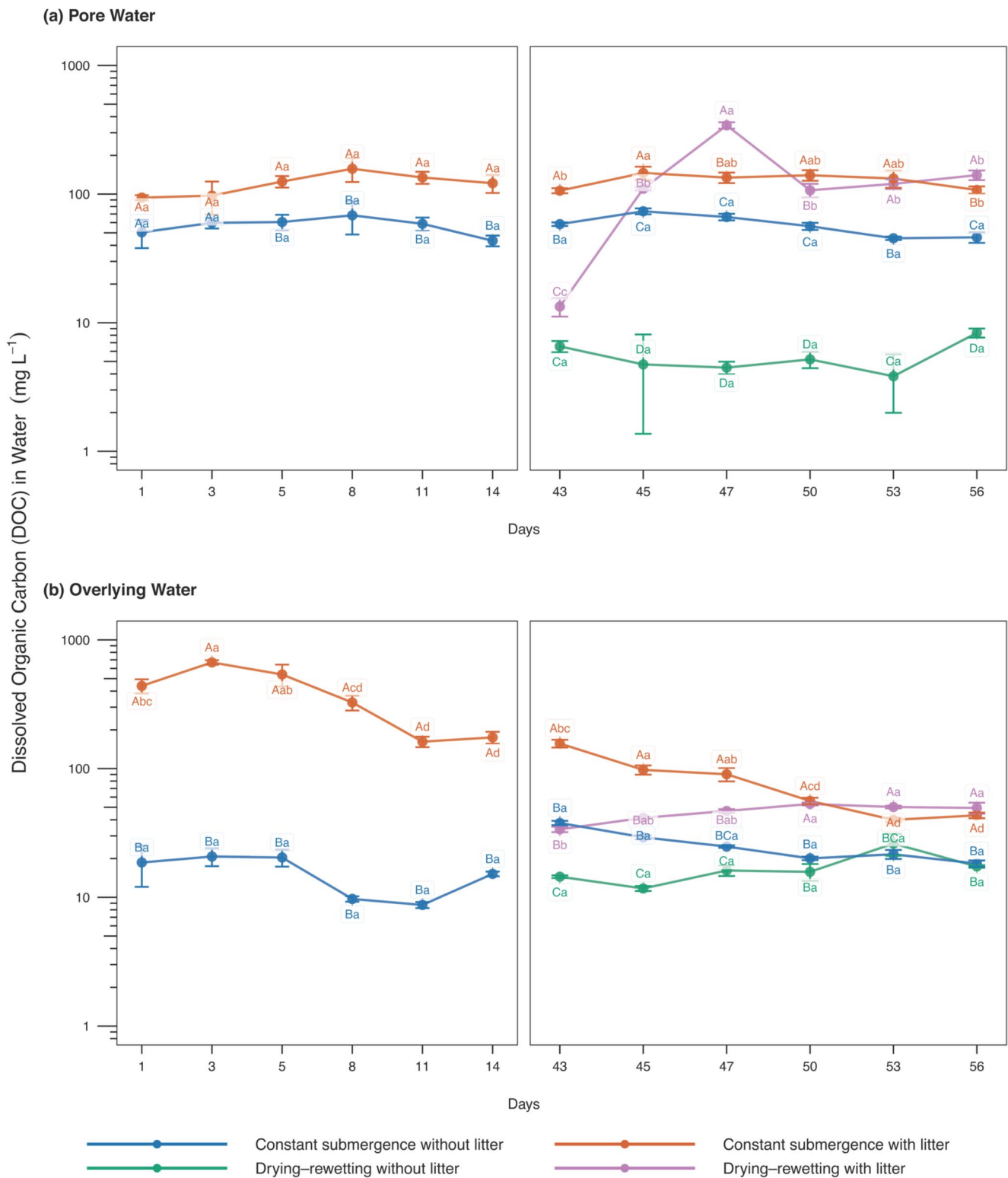


Fig. 7 Dissolved Organic Carbon (DOC) concentration in pore water (a) and overlying water (b) during Phase 1 and Phase 3. Different uppercase letters indicate significant differences among treatments at a given sampling time. Different lowercase letters indicate significant

differences over time in a given treatment ($p \leq 0.05$, $n = 4$, mean $\pm$ SE). The y-axis is plotted on a logarithmic scale to accommodate the wide range of values. Symmetric mean $\pm$ SE intervals on the original scale may appear visually asymmetric on the plot

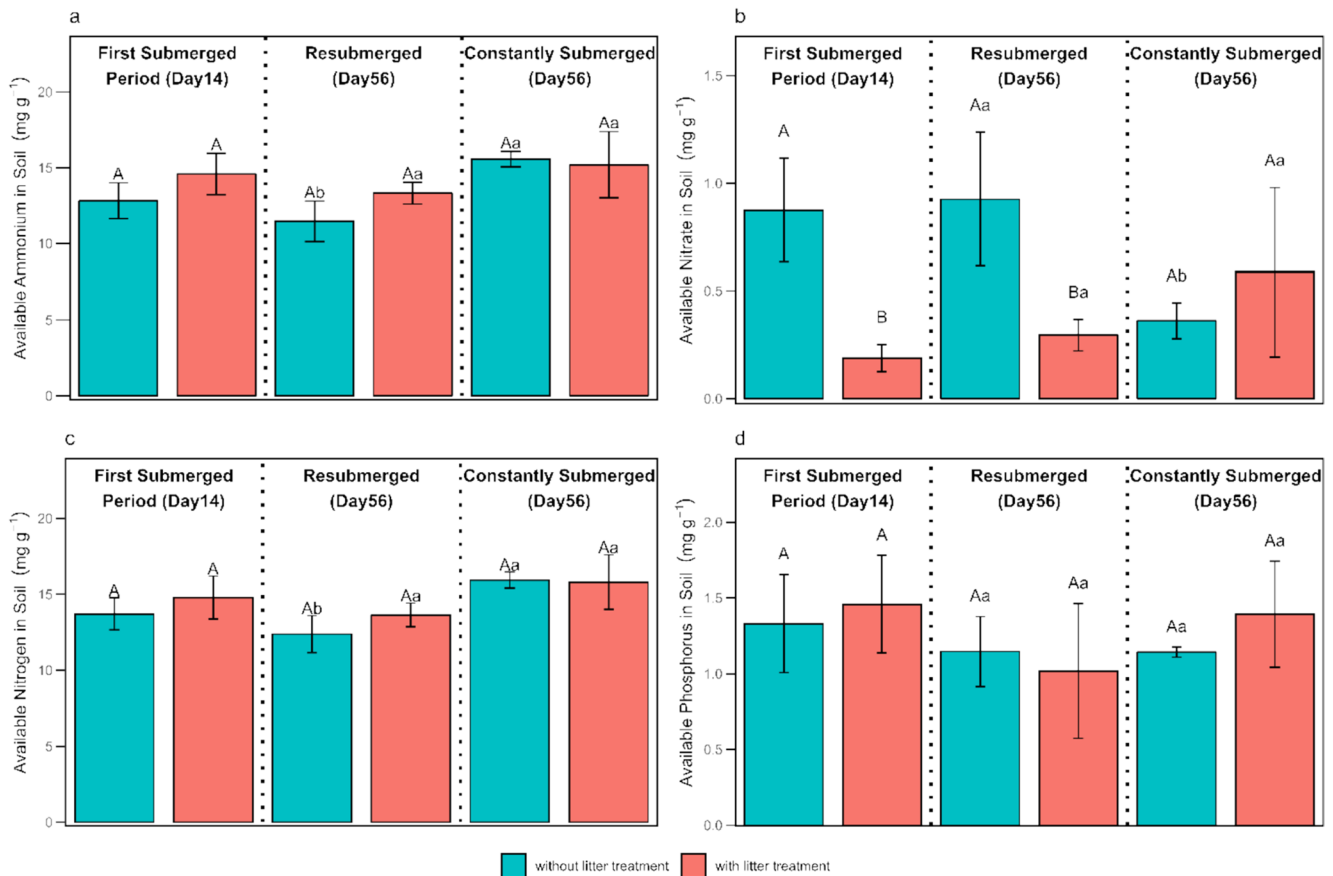


Fig. 8 Soil available (KCl extractable) ammonium (a), soil available nitrate (b), soil available nitrogen (c) and soil available phosphorus (d) under different submergence and litter treatments. Distinct uppercase letters indicate significant differences in same properties between litter treatments within the same submergence condition at a given day

(t-test: $P < 0.05$). Different lowercase letters indicate significant differences in same properties between different water treatments (constantly submerged or DRW) within the same litter treatment on harvest days (t-test: $P < 0.05$)

it remained stable at ~ 7.5 without litter. On Day 14, the pH with litter was only ~ 0.5 pH units lower than without litter. In the constantly submerged and the DRW treatments, the pH remained quite stable from Days 43 to 56 and was generally ~ 0.4 pH units lower with litter than the control without litter (Fig.S6).

Total available N in the DRW treatment without litter was $\sim 15\%$ lower than under constantly submerged conditions with litter. Soil available (KCl extractable) NH_4^+ in the DRW treatment without litter was $\sim 26\%$ lower than under constant submerged conditions without litter (Fig. 8a). Soil available NH_4^+ and available N were not affected by litter (Fig. 8a&c). Soil available NO_3^- concentration without litter was approximately four-fold higher than that with litter in both first submergence and resubmergence periods (Fig. 8b). Soil available P (Olsen extractable) was not affected by water treatments or litter addition (Fig. 8d). Redox potential measurements indicated more reducing conditions under constant submergence relative to DRW during the drying/rewetting transition (Table S2).

Discussion

This study provided insights into how a drying-rewetting cycle and litter presence shape the temporal dynamics of N, P and C across wetland soil, pore water and overlying water. It is critical to better understand how hydrological fluctuations and soil-water interactions can influence nutrient cycling processes for effective wetland management and restoration.

Nutrient Dynamics Under Constant Submergence

Under constant submergence, litter addition increased pore-water DOC and phosphorus availability and shifted dissolved N speciation relative to the no-litter control (Figs. 3, 4, 5, 6 and 7; Figs. S1-S2, S4-S5). These increases indicate diffusion of litter-derived nutrients released from decomposition in the overlying water into the soil, consistent with previous observations under constant submergence conditions (Hu et al. 2021). Constant submergence created anoxic

conditions which can promote NH_4^+ and P_i accumulation in pore water. The potential mechanism is likely through reduced nitrification and limited P sorption due to suppressed Fe(III) formation (Gardner and McCarthy 2009; Yu et al. 2017). Litter inputs enriched dissolved organic matter pools in both pore and overlying waters (Figs. S3, S5, S7), which may stimulate microbial growth and alter coupled C-N-P processing under inundation (Liang et al. 2017). Collectively, these findings indicate that organic matter inputs can intensify biogeochemical fluxes and potentially reinforce nutrient co-metabolism (e.g., linked C, N, and P transformations) under anoxic conditions. The findings also highlight the benefit of an integrated microcosm approach that considers interactions between multiple compartments (water-litter-soil).

Effect of Drying and Rewetting Cycles

In line with the first hypothesis, DRW cycles triggered pronounced shifts in N transformations across soil and water compartments (Fig.S7). Immediately after rewetting a sharp increase in pore water NH_4^+ and DON was observed. This increase likely resulted from rapid microbial activity, accelerated decomposition of organic matter, and microbial cell lysis induced by osmotic stress upon rewetting (He et al. 2025). Such conditions commonly accompany the well-known ‘Birch effect’, a phenomenon typically characterized by a transient microbial respiration following drying and subsequent rewetting. Conversely, NO_3^- levels decreased rapidly, consistent with enhanced denitrification or microbial immobilization (Venterink et al. 2002).

Interestingly, overlying water showed an initial NH_4^+ peak following rewetting (Fig.S1b), followed by a rise in NO_3^- (Fig. S2b), reflecting upward NH_4^+ diffusion from pore to overlying water and subsequent nitrification under oxic conditions (Hammerl et al. 2019). This pattern illustrates the rapid coupling of ammonification and nitrification processes during DRW events. Phosphorus dynamics were also influenced by DRW-induced redox shifts. DRW led to a net decrease in pore water P_i , likely due to Fe(II) oxidation and subsequent P sorption onto Fe(III) oxides (Schönbrunner et al. 2012; Zhang and Furman 2021).

Interactions Between Litter and DRW

Overall, the data indicate that DRW and litter inputs jointly structure cross-compartment nutrient dynamics, consistent with strong hydrology-substrate coupling during redox transitions (Fig. 2). Upon rewetting, litter treatments exhibited higher DOC and NH_4^+ concentrations (Fig.S1 and Fig. 6), indicating rapid microbial processing of labile organic inputs (Fierer and Schimel 2002). The associated decline

in pore-water NO_3^- relative to unamended columns suggests enhanced NO_3^- consumption via denitrification and/or microbial immobilisation within anaerobic microsites that develop during rewetting (Venterink et al. 2002; Scott et al. 2020). Litter addition enhanced NH_4^+ concentrations in overlying water initially, but, reduced to a similar level as in the control without litter (Fig.S1). Soil NH_4^+ did not increase (Fig.S7) which suggests this reduction was largely driven by nitrification-denitrification than soil adsorption. This interpretation is further supported by the sandy texture of the soil and its relatively low cation exchange capacity, indicating minimal NH_4^+ adsorption potential (Sommer et al. 2003).

The pronounced early DOC (and nutrient) pulse following rewetting in litter-addition treatments may reflect rapid leaching and species-specific decomposition dynamics of river red gum litter during initial wetting, which can be strongly modified by prior exposure history (Glazebrook and Robertson 1999). In addition, the high C: N ratio of the added river red gum litter likely enhanced microbial immobilization and reduced inorganic N availability in later stages, contrasting with low-C: N litter types that release more available N (Gao et al. 2016; Qu et al. 2020). In other eucalypt species (e.g., *Eucalyptus globulus*), persistent cuticular barriers and leaf oils have been shown to delay microbial colonisation and constrain early-stage breakdown, highlighting that decomposition trajectories can differ substantially even within the same genus (Canhoto and Graça 1999). In our experiment, litter residence time on the wetland margin prior to collection could not be quantified. This is important because antecedent litter ageing can reduce P content and suppress P release during subsequent rewetting. Our previous work showed that aged leaf litter had lower total P and released less P upon rewetting than fresh litter (Zhang et al. 2022). Therefore, we standardised litter by selecting visually undecomposed senesced leaves, oven-drying to constant mass, and cutting to 1–2 cm pieces to homogenise amendment size across replicates. These steps reduce among-replicate variability but may increase exposed surface area and enhance early leaching and microbial access relative to intact, naturally conditioned leaves. Consequently, the magnitude and timing of DOC and nutrient pulses observed here should be interpreted as responses under controlled, standardised litter inputs that isolate DRW mechanisms, rather than a direct quantitative representation of intact-leaf decomposition in situ. Further research examining litter types spanning a broader stoichiometric gradient and direct tracking of microbial pathways would help refine our understanding of DRW-mediated biogeochemical processes.

Phosphorus dynamics responded similarly. Both P_i and TP increased in pore water under litter+DRW treatments,

especially following the initial post-rewetting pH decline, when pH stabilised and redox conditions were rapidly shifting. Microbial decomposition of litter and redox shifts likely released Fe(III)-bound P (Sun et al. 2017). However, as mentioned above the total soil P pool remained relatively stable, indicating internal redistribution rather than net gain or loss.

Although pH was not manipulated as an experimental factor, its dynamics provide additional context for interpreting short-term nutrient partitioning. During the first submergence (Days 1–14), litter addition likely induced an initial acidification followed by partial recovery, consistent with early leaching and CO₂ production and subsequent alkalinity generation during anaerobic respiration and reductive processes (Breemen 1987; Husson 2013; Tanikawa et al. 2018). After rewetting, pore-water pH remained markedly lower than during the initial flooded phase, which may have modulated coupled N and P transformations by constraining nitrification and influencing redox-driven Fe-P sorption dynamics (Ni et al. 2023; Florea et al. 2024).

This post-rewetting acidification coincided with the strongest DRW-driven mineral-N pulse (notably NH₄⁺) and with a subsequent convergence/decline in pore-water NO₃⁻. Moreover, the rapid pH drop after rewetting is also consistent with oxidation of reduced species accumulated during the anoxic/submerged period or during drying (e.g., Fe(II) and sulfides), which can generate protons upon re-oxygenation, and therefore may not be attributable solely to NH₄⁺ oxidation (Karimian et al. 2017; Lu et al. 2022). Given the strong pH dependence of nitrifier physiology and NH₄⁺ availability, the low pore-water pH observed after rewetting could have contributed to reduced net nitrification, potentially helping to explain the persistence of NH₄⁺ in the inorganic-N time series (Ni et al. 2023). DOC exhibited its largest post-rewetting increase when litter was present, suggesting that substrate supply and hydrological disturbance primarily controlled DOC concentrations. By contrast, pH likely acted as a secondary modifier of DOC fate via pH-dependent sorption-desorption and microbial processing (Qiu et al. 2009; Spohn et al. 2022).

Conclusion

This study emphasizes the interactive influence of DRW and *Eucalyptus* species leaf litter inputs on N, P, and C dynamics in wetland ecosystems. Integrated analyses across soil, pore water, and overlying water compartments yielded several key findings.

DRW significantly enhanced N mobilization, resulting in pronounced but transient increases in NH₄⁺ and NO₃⁻ concentrations within pore water and overlying water

immediately after rewetting; however, this pulse effect diminished gradually with prolonged inundation. Litter addition amplified microbial immobilization and denitrification, substantially reducing soil NO₃⁻ concentrations and potentially mitigating N losses from wetlands. TP and P_i concentrations remained higher under constant submergence than under DRW, reflecting redox-driven P release mechanisms. Although leaf litter inputs did not significantly alter total nutrient levels in the overlying water, they considerably increased P_i and DOP proportions in pore water. Carbon dynamics were strongly influenced by the presence of leaf litter, significantly elevating DOC concentrations, particularly following DRW events. Enhanced litter decomposition during DRW cycles stimulated microbial activity, thereby influencing overall nutrient transformations.

Overall, the combined impacts of DRW and litter inputs significantly regulate nutrient availability and cycling at the aquatic-terrestrial interface. From a management perspective, leveraging DRW cycles in conjunction with high C: N litter inputs could help effectively manage inorganic nutrient availability, guiding optimized wetland restoration under variable hydrological and climatic conditions.

Supplementary Information The online version contains supplementary material available at <https://doi.org/10.1007/s13157-026-02054-7>.

Acknowledgements This work is dedicated to our co-author, Professor Petra Marschner, who sadly passed away during the preparation of this manuscript. Petra's guidance, mentorship, and passion for research continue to inspire us, and her legacy remains an integral part of our journey. We thank Dr Han Weng for his constructive comments on a draft of this manuscript.

We also extend our sincere gratitude to Andrea Paparella, Bogumila Tomczak, Brett Thomas, Dilanka Niroshan Athukorala Athukoralalage, Emily Ruth Leyden, Linwei Du, Juqi Li, and Thi Hoang Ha Truong, whose contributions were instrumental in the successful completion of this research.

Author Contributions All authors contributed to the study conception and design. Material preparation, data collection, and analysis were performed by Rui Yan, with guidance and support from Luke Mosley and Petra Marschner. The first draft of the manuscript was written by Rui Yan. Luke Mosley provided comments and approved the final manuscript. Petra Marschner contributed significantly to earlier drafts but sadly passed away before the final manuscript was completed.

Funding Open Access funding enabled and organized by CAUL and its Member Institutions. Rui Yan was financially supported by the Chinese Scholarship Council (CSC, grant No. 202108320003).

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

Declarations

Competing Interests The authors have no relevant financial or non-financial interests to disclose.

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References

- Armstrong FAJ (1963) Determination of Nitrate in Water Ultraviolet Spectrophotometry. *Anal Chem* 35:1292–1294
- Brödl D, Kaiser K, Hagedorn F (2019) Divergent patterns of carbon, nitrogen, and phosphorus mobilization in forest soils. *Front Glob Change* 2:66. <https://doi.org/10.3389/ffgc.2019.00066>
- Canhoto C, Graça MA (1999) Leaf Barriers to Fungal Colonization and Shredders (*Tipula lateralis*) Consumption of Decomposing *Eucalyptus globulus*. *Microb Ecol* 37:163–172. <https://doi.org/10.1007/s002489900140>
- Cawse PA (1967) The determination of nitrate in soil solutions by ultraviolet spectrophotometry. *Analyst* 92:311–315
- Chen J, Li H, Zhang Z et al (2020) DOC dynamics and bacterial community succession during long-term degradation of *Ulva prolifera* and their implications for the legacy effect of green tides on refractory DOC pool in seawater. *Water Res* 185:116268. <https://doi.org/10.1016/j.watres.2020.116268>
- CSIRO (2008) Water availability in the Murray. A report to the Australian Government from the CSIRO Murray-Darling Basin Sustainable Yields Project. CSIRO, Australia
- Davidson NC, van Dam AA, Finlayson CM, McInnes RJ (2019) Worth of wetlands: revised global monetary values of coastal and inland wetland ecosystem services. *Mar Freshw Res* 70:1189–1194. <http://doi.org/10.1071/MF18391>
- de Vicente I (2021) Biogeochemistry of Mediterranean Wetlands: A Review about the Effects of Water-Level Fluctuations on Phosphorus Cycling and Greenhouse Gas Emissions. *Water* 13:1510. <https://doi.org/10.3390/w13111510>
- Ding Y-D, Song C-C, Chen G-J et al (2021) Effects of long-term nitrogen addition on dissolved organic matter characteristics in a temperate wetland of Northeast China. *Ecotoxicol Environ Saf* 226:112822. <https://doi.org/10.1016/j.ecoenv.2021.112822>
- Ebina J, Tsutsui T, Shirai T (1983) Simultaneous determination of total nitrogen and total phosphorus in water using peroxodisulfate oxidation. *Water Res* 17:1721–1726
- Edzwald JK, Becker WC, Wattier KL (1985) Surrogate parameters for monitoring organic matter and THM precursors. *Journal-American Water Works Assoc* 77:122–132
- Esslemont G, Maher W, Ford P, Lawrence I (2007) Riparian Plant Material Inputs to the Murray River, Australia. *J Environ Qual* 36:963–974. <https://doi.org/10.2134/jeq2006.0318>
- Fierer N, Schimel JP (2002) Effects of drying–rewetting frequency on soil carbon and nitrogen transformations. *Soil Biol Biochem* 34:777–787. [https://doi.org/10.1016/S0038-0717\(02\)00007-X](https://doi.org/10.1016/S0038-0717(02)00007-X)
- Finlayson CM, Davis JA, Gell PA et al (2013) The status of wetlands and the predicted effects of global climate change: the situation in Australia. *Aquat Sci* 75:73–93. <https://doi.org/10.1007/s00027-011-0232-5>
- Florea AF, Heckrath G, Zak DH et al (2024) Phosphorus release from rewetted agricultural peat soils varies strongly in dependence of the phosphorus resorption capacity. *Geoderma* 441:116739. <https://doi.org/10.1016/j.geoderma.2023.116739>
- Gao J, Feng J, Zhang X et al (2016) Drying–rewetting cycles alter carbon and nitrogen mineralization in litter-amended alpine wetland soil. *CATENA* 145:285–290. <https://doi.org/10.1016/j.catena.2016.06.026>
- Gao D, Bai E, Li M et al (2020) Responses of soil nitrogen and phosphorus cycling to drying and rewetting cycles: A meta-analysis. *Soil Biol Biochem* 148:107896. <https://doi.org/10.1016/j.soilbio.2020.107896>
- Gardner WS, McCarthy MJ (2009) Nitrogen dynamics at the sediment–water interface in shallow, sub-tropical Florida Bay: why denitrification efficiency may decrease with increased eutrophication. *Biogeochemistry* 95:185–198. <https://doi.org/10.1007/s10533-009-9329-5>
- Gee GW, Or D (2002) 2.4 Particle-size analysis. *Methods Soil Anal Part 4 Phys Methods* 5:255–293. <https://doi.org/10.2136/sssabookser5.4.c12>
- George ER, David JL (2010) *Soil Chemical Methods - Australasia*. CSIRO Publishing
- Gibbs MS, Wallace T, Mosley LM (2022) Constraining organic matter composition and dynamics as a dominant driver of hypoxic black-water risk during river Murray floodplain inundation. *Hydrol Process* 36:e14529. <https://doi.org/10.1002/hyp.14529>
- Glazebrook HS, Robertson AI (1999) The effect of flooding and flood timing on leaf litter breakdown rates and nutrient dynamics in a river red gum (*Eucalyptus camaldulensis*) forest. *Aust J Ecol* 24:625–635. <https://doi.org/10.1046/j.1442-9993.1999.00992.x>
- Gómez R, Arce MI, Sánchez JJ (2012) The effects of drying on sediment nitrogen content in a Mediterranean intermittent stream: a microcosms study. *Hydrobiologia* 679:43–59. <https://doi.org/10.1007/s10750-011-0854-6>
- Gu B, Mehlhorn TL, Liang L, McCarthy JF (1996) Competitive adsorption, displacement, and transport of organic matter on iron oxide: I. Competitive adsorption. *Geochim Cosmochim Acta* 60:1943–1950. [https://doi.org/10.1016/0016-7037\(96\)00059-2](https://doi.org/10.1016/0016-7037(96)00059-2)
- Gu S, Gruau G, Malique F et al (2018) Drying/rewetting cycles stimulate release of colloidal-bound phosphorus in riparian soils. *Geoderma* 321:32–41. <https://doi.org/10.1016/j.geoderma.2018.01.015>
- Hammerl V, Kastl E-M, Schlöter M et al (2019) Influence of rewetting on microbial communities involved in nitrification and denitrification in a grassland soil after a prolonged drought period. *Sci Rep* 9:2280. <https://doi.org/10.1038/s41598-018-38147-5>
- Hanson WC (1950) The photometric determination of phosphorus in fertilizers using the phosphovanado-molybdate complex. *J Sci Food Agric* 1:172–173. <https://doi.org/10.1002/jsfa.2740010604>
- He X, Yan W, Chen X et al (2025) The transition from macrophyte-dominated to algae-dominated lake systems enhances arsenic release from sediments. *Water Res* 276:123233. <https://doi.org/10.1016/j.watres.2025.123233>
- Hömborg A, Matzner E (2018) Effects of drying and rewetting on soluble phosphorus and nitrogen in forest floors: An experiment with undisturbed columns. *J Plant Nutr Soil Sci* 181:177–184. <https://doi.org/10.1002/jpln.201700380>
- Homyak PM, Allison SD, Huxman TE et al (2017) Effects of Drought Manipulation on Soil Nitrogen Cycling: A Meta-Analysis. *J Geophys Res Biogeosciences* 122:3260–3272. <https://doi.org/10.1002/2017JG004146>
- Hu X, Xie T, Arif M et al (2021) Response of Annual Herbaceous Plant Leaching and Decomposition to Periodic Submergence in Mega-Reservoirs: Changes in Litter Nutrients and Soil Properties for Restoration. *Biology* 10:1141. <https://doi.org/10.3390/biology10111141>

- Husson O (2013) Redox potential (Eh) and pH as drivers of soil/plant/microorganism systems: a transdisciplinary overview pointing to integrative opportunities for agronomy. *Plant Soil* 362:389–417. <https://doi.org/10.1007/s11104-012-1429-7>
- Karanfil T, Erdogan I, Schlautman MA (2003) Selecting filter membranes for measuring DOC and UV254. *Journal-American Water Works Assoc* 95:86–100. <https://doi.org/10.1002/j.1551-8833.2003.tb10317.x>
- Karimian N, Johnston SG, Burton ED (2017) Acidity generation accompanying iron and sulfur transformations during drought simulation of freshwater re-flooded acid sulfate soils. *Geoderma* 285:117–131. <https://doi.org/10.1016/j.geoderma.2016.09.030>
- Kinsman-Costello LE, Hamilton SK, O'Brien JM, Lennon JT (2016) Phosphorus release from the drying and reflooding of diverse shallow sediments. *Biogeochemistry* 130:159–176. <https://doi.org/10.1007/s10533-016-0250-4>
- Leitner S, Homyak PM, Blankinship JC et al (2017) Linking NO and N₂O emission pulses with the mobilization of mineral and organic N upon rewetting dry soils. *Soil Biol Biochem* 115:461–466. <https://doi.org/10.1016/j.soilbio.2017.09.005>
- Li C, Wang Y, Yi Y et al (2024) A review of reservoir carbon Cycling: Key Processes, influencing factors and research methods. *Ecol Indic* 166:112511. <https://doi.org/10.1016/j.ecolind.2024.112511>
- Li L, Tan J, Yang G et al (2025) Regulatory Mechanisms of Tannins on the Decomposition Rate of Mixed Leaf Litter in Submerged Environments. *Plants* 14:3064. <https://doi.org/10.3390/plants14193064>
- Liang C, Schimel JP, Jastrow JD (2017) The importance of anabolism in microbial control over soil carbon storage. *Nat Microbiol* 2:1–6. <https://doi.org/10.1038/nmicrobiol.2017.105>
- Lopez-Sangil L, Hartley IP, Rovira P et al (2018) Drying and rewetting conditions differentially affect the mineralization of fresh plant litter and extant soil organic matter. *Soil Biol Biochem* 124:81–89. <https://doi.org/10.1016/j.soilbio.2018.06.001>
- Lu H, Li K, Nkoh JN et al (2022) Effects of pH variations caused by redox reactions and pH buffering capacity on Cd(II) speciation in paddy soils during submerging/drainage alternation. *Ecotoxicol Environ Saf* 234:113409. <https://doi.org/10.1016/j.ecoenv.2022.113409>
- Luo P, Tong X, Liu F et al (2020) Nutrients release and greenhouse gas emission during decomposition of *Myriophyllum aquaticum* in a sediment-water system. *Environ Pollut* 260:114015. <https://doi.org/10.1016/j.envpol.2020.114015>
- Marschner P (2021) Processes in submerged soils – linking redox potential, soil organic matter turnover and plants to nutrient cycling. *Plant Soil* 464:1–12. <https://doi.org/10.1007/s11104-021-05040-6>
- Matthews KE, Facelli JM, Cavagnaro TR (2023) Response of soil microbial community structure, carbon and nitrogen cycling to drying and rewetting. *Appl Soil Ecol* 192:105099. <https://doi.org/10.1016/j.apsoil.2023.105099>
- McPhan L, Capon S, Bond N (2022) Reconnecting River Country – Floodplain Vegetation Condition Predictive Modelling; Part 1: Murray River floodplain (CFE Publication #274). NSW Department of Planning and Environment, New South Wales, Australia
- McKenzie HA, Wallace HS (1954) The Kjeldahl determination of nitrogen: a critical study of digestion conditions-temperature, catalyst, and oxidizing agent. *Aust J Chem* 7:55–70. <https://doi.org/10.1071/CH9540055>
- Miranda KM, Espey MG, Wink DA (2001) A rapid, simple spectrophotometric method for simultaneous detection of nitrate and nitrite. *Nitric Oxide* 5:62–71. <https://doi.org/10.1006/niox.2000.0319>
- Najera F, Dippold MA, Boy J et al (2020) Effects of drying/rewetting on soil aggregate dynamics and implications for organic matter turnover. *Biol Fertil Soils* 56:893–905. <https://doi.org/10.1007/s00374-020-01469-6>
- Ni G, Leung PM, Daebeler A et al (2023) Nitrification in acidic and alkaline environments. *Essays Biochem* 67:753–768. <https://doi.org/10.1042/EBC20220194>
- Ohno T, Zibilske LM (1991) Determination of Low Concentrations of Phosphorus in Soil Extracts Using Malachite Green. *Soil Sci Soc Am J* 55:892–895. <https://doi.org/10.2136/sssaj1991.03615995005500030046x>
- Paul S, Grocke S, Baker AK et al (2011) Assessment of Acid Sulfate Soil Materials (Phase 2) Teal Flat Hut wetland. CSIRO, Australia
- Petrakis S, Seyfferth A, Kan J et al (2017) Influence of experimental extreme water pulses on greenhouse gas emissions from soils. *Biogeochemistry* 133:147–164. <https://doi.org/10.1007/s10533-017-0320-2>
- Pezeshki SR, DeLaune RD (2012) Soil Oxidation-Reduction in Wetlands and Its Impact on Plant Functioning. *Biology* 1:196–221. <https://doi.org/10.3390/biology1020196>
- Qiu Y, Xiao X, Cheng H et al (2009) Influence of Environmental Factors on Pesticide Adsorption by Black Carbon: pH and Model Dissolved Organic Matter. In: ACS Publ. <https://pubs.acs.org/doi/pdf/10.1021/es900573d>. Accessed 16 Jan 2026
- Qu W, Han G, Eller F et al (2020) Nitrogen input in different chemical forms and levels stimulates soil organic carbon decomposition in a coastal wetland. *CATENA* 194:104672. <https://doi.org/10.1016/j.catena.2020.104672>
- R Core Team (2024) R: A Language and Environment for Statistical Computing
- Rodríguez A, Durán J, Rey A et al (2019) Interactive effects of forest die-off and drying-rewetting cycles on C and N mineralization. *Geoderma* 333:81–89. <https://doi.org/10.1016/j.geoderma.2018.07.003>
- Sawada K, Funakawa S, Toyota K, Kosaki T (2015) Potential nitrogen immobilization as influenced by available carbon in Japanese arable and forest soils. *Soil Sci Plant Nutr* 61:917–926. <https://doi.org/10.1080/00380768.2015.1075364>
- Schönbrunner IM, Preiner S, Hein T (2012) Impact of drying and reflooding of sediment on phosphorus dynamics of river-floodplain systems. *Sci Total Environ* 432:329–337. <https://doi.org/10.1016/j.scitotenv.2012.06.025>
- Scott B, Baldwin AH, Ballantine K et al (2020) The role of organic amendments in wetland restorations. *Restor Ecol* 28:776–784. <https://doi.org/10.1111/rec.13179>
- Sommer SG, Générmont S, Cellier P et al (2003) Processes controlling ammonia emission from livestock slurry in the field. *Eur J Agron* 19:465–486. [https://doi.org/10.1016/S1161-0301\(03\)00037-6](https://doi.org/10.1016/S1161-0301(03)00037-6)
- Spohn M, Diáková K, Aburto F et al (2022) Sorption and desorption of organic matter in soils as affected by phosphate. *Geoderma* 405:115377. <https://doi.org/10.1016/j.geoderma.2021.115377>
- Sun D, Li K, Bi Q et al (2017) Effects of organic amendment on soil aggregation and microbial community composition during drying-rewetting alternation. *Sci Total Environ* 574:735–743. <https://doi.org/10.1016/j.scitotenv.2016.09.112>
- Tanikawa T, Fujii S, Sun L et al (2018) Leachate from fine root litter is more acidic than leaf litter leachate: A 2.5-year laboratory incubation. *Sci Total Environ* 645:179–191. <https://doi.org/10.1016/j.scitotenv.2018.07.038>
- van Breemen N (1987) Effects of redox processes on soil acidity. *Neth J Agric Sci* 35:271–279. <https://doi.org/10.18174/njas.v35i3.16724>
- Venterink HO, Davidsson TE, Kiehl K, Leonardson L (2002) Impact of drying and re-wetting on N, P and K dynamics in a wetland soil. *Plant Soil* 243:119–130. <https://doi.org/10.1023/A:1019993510737>
- Vigo CN, Oclocho-Garcia FE, Iliquin Trigoso D, Oliva-Cruz M (2024) Influence of *Eucalyptus globulus* plantations on soil characteristics

- at different altitudinal levels. *Trees People* 18:100677. <https://doi.org/10.1016/j.tfp.2024.100677>
- Walkley A, Black IA (1934) An examination of the Degtjareff method for determining soil organic matter, and a proposed modification of the chromic acid titration method. *Soil Sci* 37:29–38. <https://doi.org/10.1097/00010694-193401000-00003>
- Wang C, Guo J, Zhang W et al (2021) Drying-rewetting changes soil phosphorus status and enzymatically hydrolysable organic phosphorus fractions in the water-level fluctuation zone of Three Gorges reservoir. *CATENA* 204:105416. <https://doi.org/10.1016/j.catena.2021.105416>
- White JR, Reddy KR (2009) Biogeochemical dynamics I: Nitrogen cycling in wetlands. *Wetl Handb* 2:213–227. <https://doi.org/10.1002/9781444315813.ch9>
- Wilke B-M (2005) Determination of chemical and physical soil properties. *Monit Assess Soil Bioremediat* 47–95. https://doi.org/10.1007/3-540-28904-6_2
- Willis RB, Montgomery ME, Allen PR (1996) Improved method for manual, colorimetric determination of total Kjeldahl nitrogen using salicylate. *J Agric Food Chem* 44:1804–1807. <https://doi.org/10.1021/jf950522b>
- Xiang S-R, Doyle A, Holden PA, Schimel JP (2008) Drying and rewetting effects on C and N mineralization and microbial activity in surface and subsurface California grassland soils. *Soil Biol Biochem* 40:2281–2289. <https://doi.org/10.1016/j.soilbio.2008.05.004>
- Xu X, Yin L, Duan C, Jing Y (2016) Effect of N addition, moisture, and temperature on soil microbial respiration and microbial biomass in forest soil at different stages of litter decomposition. *J Soils Sediments* 16:1421–1439. <https://doi.org/10.1007/s11368-015-1331-z>
- Yu J, Ding S, Zhong J et al (2017) Evaluation of simulated dredging to control internal phosphorus release from sediments: Focused on phosphorus transfer and resupply across the sediment-water interface. *Sci Total Environ* 592:662–673. <https://doi.org/10.1016/j.scitotenv.2017.02.219>
- Zhang Z, Furman A (2021) Soil redox dynamics under dynamic hydrologic regimes - A review. *Sci Total Environ* 763:143026. <https://doi.org/10.1016/j.scitotenv.2020.143026>
- Zhang X, Smernik R, Doolette A et al (2022) Phosphorus speciation and release from different plant litters on a River Murray (Australia) floodplain. *Plant Soil* 471:141–156. <https://doi.org/10.1007/s11104-021-05197-0>
- Zhang X, Chen F, Yan D et al (2024) Effects of wet-dry alternation on organic phosphorus dynamics and sediment characteristics in the intertidal zone of Nansi Lake. *Ecotoxicol Environ Saf* 281:116668. <https://doi.org/10.1016/j.ecoenv.2024.116668>

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