



Direct and indirect pathways of land management effects on wetland plant litter decomposition



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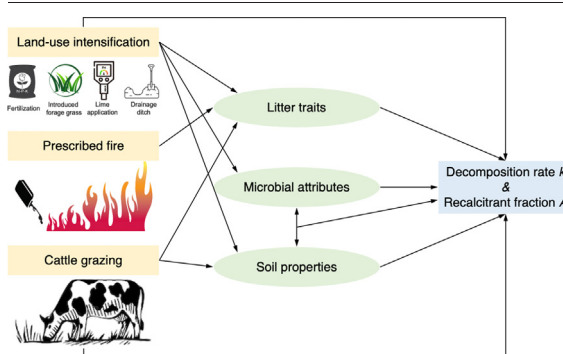
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HIGHLIGHTS

- Upland agricultural intensification stimulated wetland litter decomposition by increasing rates and completeness.
- Prescribed fire showed inhibitory effects on wetland litter decomposition.
- Cattle grazing affected decomposition depended on land-use intensity, and its impacts were weaker than intensification or fire.
- Land management affected decomposition through altering litter and soil and microbial attributes, and their interactions.
- Litter and soil variables strongly predicted decomposition, but specific best predictors for the rates and completeness varied.

GRAPHICAL ABSTRACT



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ABSTRACT

Litter decomposition is a fundamental process underpinning multiple ecosystem services. Despite a long history of research on decomposition, direct and indirect effects of multiple interactive land management on wetland decomposition yet remain less well understood. Here, we used a long-term whole-ecosystem wetland experiment in south-central Florida to investigate interactive effects of land-use intensification, cattle grazing and prescribed fire on in situ wetland plant litter decomposition. We further examined the direct and indirect pathways of land management effects on litter decomposition through changes in associated litter traits, soil properties, and soil microbial attributes using structural equation models. We used the litterbag technique that quantifies decomposition rates (k -values) and recalcitrant fractions (A -values). Our results showed that land-use intensification increased k -values in ungrazed wetlands and decreased k -values in grazed wetlands, but consistently reduced A -values regardless of other treatments. Prescribed fire individually suppressed litter decomposition by reducing k and increasing A . Further, these effects occurred through altering litter, soil, and microbial properties. Our results revealed that litter traits and soil properties were the first two strongest factors in determining wetland decomposition processes. Particularly, litter P and Mg contents and soil P and K contents were the best predictors for k , while litter Ca and lignin contents and soil pH, N and water content best predicted A . Moreover, microbial traits exhibited interactive effects with litter and soil properties to affect wetland litter decomposition. Our research suggests that cattle grazing could buffer against stimulating effect of land-use intensification on decomposition rates and thus avoid nutrient releases pulses. Our study further indicates that land-use intensification and fire suppression in subtropical wetlands could promote organic matter depletion and

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thus nutrient loss, highlighting the need to reduce anthropogenic disturbances to natural wetlands to maintain their capacity for providing associated regulating and supporting services.

1. Introduction

Geographically isolated wetlands provide critical ecosystem services such as flood protection, carbon sequestration, nutrient retention, water purification, and biodiversity support (Cohen et al., 2016; Janse et al., 2019; McClain et al., 2003; Whigham, 1999). In tropical and subtropical regions, numerous isolated seasonal wetlands are embedded across agricultural landscapes (Mitsch et al., 2010). Although these wetlands do not occupy a large proportion of the land surface, they are widely distributed across the landscapes, and function as hotspots of ecosystem services and indispensable environmental buffers between agricultural lands and natural areas (Janse et al., 2019; Whigham, 1999). However, as human population grows and demands for food and agricultural products increase, these wetlands are facing constant and emerging threats from anthropogenic activities, including intensified agricultural land uses, expansion of livestock grazing, and associated agroecosystem management practices (Alexandratos and Bruinsma, 2012; Asner et al., 2004). These agricultural and land management activities, while fulfilling basic food demands, have inevitably transformed wetlands in multiple ways, such as increased nutrient loading and soil disturbances, altered hydrologic regimes, species shifts in biological communities, and reduced habitat heterogeneity and biodiversity (Reid et al., 2005; Tilman, 1999). All these biotic and abiotic changes induced by land management are likely to affect organic matter decomposition – a fundamental ecological process that regulates biogeochemical cycling, energy flows, and underpins delivery of multiple ecosystem services. Therefore, we need a comprehensive understanding of how organic matter decomposition in these wetlands respond to individual and interactive agricultural management practices, to inform effective and sustainable land management and ensure the long-term provision of vital ecosystem services in agriculture-dominated landscapes.

The processes of organic matter decomposition determine various essential ecosystem functions and services, including organic matter accumulation, greenhouse gas mitigation, soil formation, and soil health maintenance (Richardson and Hanna, 2021). In terrestrial ecosystems, >50 % of net primary production is returned to soil through the decomposition of plant litter (Wardle et al., 2004). Therefore, understanding how anthropogenic activities alter litter decomposition has major implications for predicting carbon budgets, and achieving sustainable land management and multifunctional agriculture (Zuazo et al., 2011). While there has been a long history of research on litter decomposition, effects of agricultural land management on decomposition have been inconsistent with complex responses to different and sometimes interacting management practices. For example, nitrogen (N) and phosphorous (P) additions, which are common agricultural intensification practices, were reported to stimulate leaf decomposition of trees and shrubs in a Brazilian savanna (Jacobson et al., 2011) and subtropical/tropical forests (Hobbie, 2000), inhibit litter decomposition in forests under high levels of N deposition (Zheng et al., 2017), and show no effect on litter decomposition in corn production systems (Grandy et al., 2013). Similarly, livestock grazing exhibited varied effects on litter decomposition in different systems. For instance, cattle and sheep grazing reduced decomposition of tree leaf litter in an Australian temperate woodland (Lindsay and Cunningham, 2009), but enhanced grasses and forbs litter decomposition in a semi-arid grassland (Naeem et al., 2021). Thus, it is critical to investigate indirect pathways in which multiple agriculture land use and management practices affect litter decomposition.

Prior research has demonstrated that litter decomposition is primarily controlled by litter quality, environmental conditions, and decomposing organisms (Chapin et al., 2002; Swift et al., 1979). Litter quality is commonly indicated by concentrations of N, P, and lignin, and ratios of C:N and N:lignin in plant tissue (Wardle et al., 2002). In addition, other plant functional traits,

such as concentrations of essential macronutrients (e.g., potassium (K), calcium (Ca), magnesium (Mg)), have been increasingly recognized as important for predicting decomposition processes (Makkonen et al., 2012; Yue et al., 2021). However, the direction and magnitude of effects of litter traits on decomposition vary across study systems (Krishna and Mohan, 2017; Wang et al., 2013; Wardle et al., 2002), probably due to complex interactions with decomposers and environmental conditions that are responsive to management practices (García-Palacios et al., 2013; Ge et al., 2013). For example, high-quality litter (e.g., high N but low lignin content) usually decomposes faster or more completely than low-quality litter (Chapin et al., 2002; Swift et al., 1979), because microbial mining on complex polymers (e.g., lignin) requires synthesis of multiple extracellular enzymes that are metabolically costly (Ferreira et al., 2015). However, increasing nutrient availability (e.g., through fertilization) might stimulate activity of microbial decomposers, such as by promoting fungal biomass accrual and sporulation (Gulis et al., 2006), and thus offset negative effects of high lignin content on litter decomposition. Moreover, litter quality could affect the composition of in situ soil microbial communities, which in turn drives its decomposition (Yan et al., 2018). Nevertheless, the specific role of soil microbial composition and structure in response to land management and consequences for litter decomposition has not been thoroughly investigated (Ge et al., 2013; Matulich and Martiny, 2015). In particular, compared to previous studies that focused on effects of individual factors (Ferreira et al., 2015; Krishna and Mohan, 2017; Yarwood, 2018), effects of multiple land management practices (which are interactive and often operating in concert) on litter decomposition through interactions among litter traits, soil properties, and soil microbial communities yet remain less well understood (Krishna and Mohan, 2017; Yarwood, 2018).

In this research, we used a field experiment to investigate plant litter decomposition in isolated seasonal wetlands embedded in managed grasslands in south-central Florida, USA (Fig. 1). Our research is built upon a long-term whole-ecosystem and factorial wetland experiment (Boughton et al., 2016) that manipulates effects of land-use intensification, cattle grazing, and prescribed fire, all of which are widely adopted agricultural management and critical global change drivers for agroecosystems (Steffen et al., 2006). Specifically, we ask: (1) How do land-use intensification, cattle grazing, and prescribed fire interact to affect decomposition in subtropical wetlands embedded in agroecosystems? (2) Among litter, soil, and microbial attributes, what are the most powerful predictors for litter decomposition? (3) What are the direct and indirect pathways through land-use intensification and management practices affect wetland decomposition? To answer these questions, we used the litterbag technique (Hackney and De La Cruz, 1980) with in-situ dominant plant species to measure decomposition processes that include quantification of decomposition rate k and recalcitrant fraction A . We also measured litter traits of dominant plant species, wetland soil properties, and soil bacterial and fungal communities to elucidate the mechanisms of litter decomposition processes in response to agricultural land-use intensification and management practices. Overall, we hypothesized that land-use intensification and cattle grazing could promote wetland litter decomposition that hinders accumulations of stable organic matter. However, management practices that mimic natural disturbances, such as short-interval prescribed fire, would suppress litter decomposition.

2. Materials and methods

2.1. Study area

Our research was conducted at Archbold Biological Station's Buck Island Ranch (BIR) in Lake Placid, Florida, USA (27°09' N, 81°11' W). BIR is a 4336-ha commercial cow-calf ranch and is part of the Archbold-University

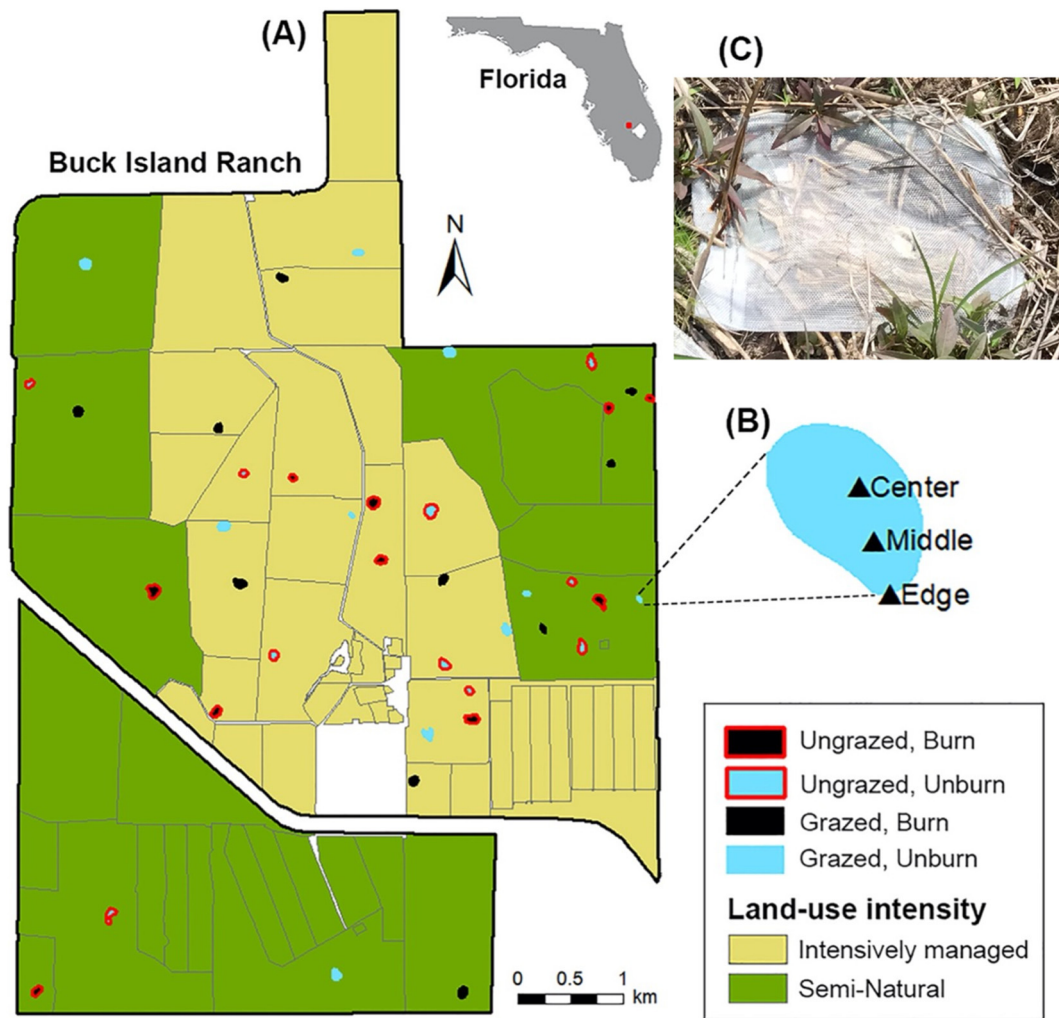


Fig. 1. (A) Map of the 40 wetlands in a whole-ecosystem experiment at Archbold Biological Station Buck Island Ranch, Lake Placid, Florida, USA. Intensively managed (IM) pastures are shown in yellow, and semi-natural (SN) pastures are shown in green. Fenced (ungrazed) wetlands are illustrated with red outlines, and grazed wetlands with no outlines. Wetlands with prescribed fire treatment are shown in black color, and wetlands with no fire treatment are filled with light blue color. (B) A set four identical litterbags were deployed at sampling sites – center, middle, and edge – for each wetland to capture the spatial variation along the hydrological gradient. (C) Photo is shown as an example of litterbag deployed in the field.

of Florida Long-Term Agroecosystem Research (LTAR) site (Fig. 1). The region has a humid subtropical climate consisting of a warm dry season (November–May) and a hot wet season (June–October), with annual average rainfall of 136-cm, 75 % of which typically falls in the wet season. Average minimum and maximum daily temperatures are 15.9 °C and 29.0 °C, respectively. BIR pastures have been managed under two land-use intensities that are exemplary of local ranching systems and large subtropical South American regions (Swain et al., 2007): intensively managed (IM), and semi-natural (SN) pastures, along with seasonal wetlands embedded in these pastures (Fig. 1). Intensively managed pastures have been heavily drained through constructed ditches, abundantly planted with forage grasses (mainly *Paspalum notatum*), and experienced high grazing intensity, with 391 average animal use days (AUD) per hectare. Moreover, a portion of BIR's IM pastures are regularly limed, N-fertilized every 1–2 years, and until 1987 fertilized with P and K. In contrast, semi-natural pastures were lightly drained, sparsely planted with forage grasses, experienced low grazing intensity (with an average AUD of 178 per hectare) and were never limed or fertilized.

On BIR, there are >600 small isolated seasonal wetlands, totaling 12 % of the area of the ranch (Swain et al., 2013), which are important landscape elements and influenced by upland agricultural land management. Wetlands embedded in IM pastures had higher soil nutrients (Ho et al., 2018) but lower diversity in plant, macroinvertebrate, and ectothermic vertebrates'

communities (Boughton et al., 2016; Medley et al., 2015) than their counterparts in SN pastures. For convenience, in subsequent sections, we will term wetlands embedded in IM or SN pastures as IM or SN wetlands. In addition to upland intensification, grazing and prescribed fire are another two most common management and disturbances in these tropical/subtropical agroecosystems, which also have shown significant effects on wetland soil nutrients, water quality, and biodiversity from past studies (Boughton et al., 2016; Ho et al., 2018; Jansen et al., 2019). For example, research from the same study region found that cattle grazing and prescribed fire tended to promote P sequestration in IM wetlands (Ho et al., 2018). However, for plant communities, grazing has been found to increase richness of short graminoids and forbs, such as *Bacopa caroliniana*, whereas fire reduced richness of graminoids (Boughton et al., 2016).

2.2. Experimental design

To investigate effects of land management on organic matter decomposition, we conducted a litterbag decomposition experiment within an existing whole-ecosystem, long-term wetland experiment that was initialized in 2007. Details on the experimental design can be found in Boughton et al. (2016) and Guo et al. (2021). In brief, a $2 \times 2 \times 2$ complete factorial design with a randomized complete block was setup using forty similar-size wetlands distributed across BIR (0.5–1.5 ha) (Fig. 1A). The eight treatment

combinations are the crossed presence/absence of cattle grazing and prescribed fire in IM and SN wetlands, respectively, resulting in five replicates across five blocks for each unique treatment combination. For grazed treatment implementation, cattle activities were tracked by recording when cattle were present in pastures that contained experimental wetlands; while for ungrazed (fenced) treatment, cattle enclosures were installed in 2007. For burned treatment, prescribed fire was implemented during dry seasons in 2008, 2011, 2013–2014, and 2017–2018 with return intervals mimicking natural fire regimes.

For each wetland, we selected three sites (i.e., center, middle, and edge) along the hydrological gradient to capture the variation in decomposition within a wetland (Fig. 1B). The center site was the lowest elevation point of the wetland; the edge site was estimated as one of the highest and driest point of the wetland and determined by the longest distance between center site and wetland margin; and the middle site was the midpoint with equal distances to the center and edge sites (Guo et al., 2021). The line formed by these three sites shows clear transitions in vegetation well aligning with the hydrological gradient (Boughton et al., 2010).

2.3. Litterbag technique to quantify decomposition

We employed the litterbag technique (Hackney and De La Cruz, 1980) to measure decomposition processes of aboveground in situ plant litter in wetlands. Specifically, large 30 × 20 cm nylon bags with 2-mm mesh size were used as litterbags because we wanted to avoid manual fragmentation of plant litter and allow access of macroinvertebrate decomposers. Each litterbag was filled with 10-g dried plant materials representative of in situ litter. Composition of plant materials used in litterbag for each wetland type is shown in Table 1, as determined based on most recent plant surveys in these forty wetlands in 2018 (unpublished data from E.H. Boughton). Plant species with >10 % frequency were selected, and their relative abundance in litterbags matched with those in the field (Table 1). Plant materials were collected from one representative wetland within each of the eight treatment types at the end of the growing season in October 2018. Collected plants were carefully rinsed to remove sediment then oven-dried at 40 °C for 72 h.

Four litterbags were placed for each sampling site on the sediment surface and secured using landscape pins (Fig. 1C) in December 2018. A total of 480 litterbags (40 wetlands × 3 sites × 4 litterbags per site) were deployed. In grazed wetlands, we constructed a 1-m diameter circular fence with chicken wire and T-posts surrounding deployed litterbags per

sampling site to exclude direct cattle trampling. We then retrieved one random litterbag from each site at 4 intervals in January, March, June, and December 2019 (i.e., 1, 3, 6, and 12 months after deployment). Retrieved litterbags were sealed in plastic Ziplock bags during transportation, then stored at 4 °C in the laboratory until litter were gently rinsed to remove adhering sediments and oven-dried at 60 °C for 72 h. Remaining litter was weighed, and the proportion of mass remaining (MR) was calculated by dividing by the initial weight. The decomposition rate k (i.e., decomposition constant per day) was estimated using the single negative exponential regression (Eq. (1)) as follows:

$$MR(t) = e^{-kt} \quad (1)$$

where $MR(t)$ is the proportion of litter mass remaining after t days, and k is the decomposition constant. In addition, we estimated the recalcitrant fraction A using MR and negative asymptotic regressions (Eq. (2)):

$$MR(t) = A + (1 - A)e^{-kt} \quad (2)$$

where A is the asymptote, or the fraction of recalcitrant plant litter. A higher recalcitrant fraction of plant litter indicates a lower completeness of decomposition. Two different regressions were used for estimating k and A , because studies have shown that the single negative exponential regression provided a more accurate estimation of actual decomposition rate (Stagg et al., 2018).

2.4. Characterizing litter traits, soil properties and microbial communities

A subsample of initial dried plant litter was ground and oven-dried at 85 °C for measurement of initial litter traits, including concentrations of carbon (C), N, P, K, Ca, Mg, and lignin, as well as the ratio of C to N (C/N), C to P (C/P), and N to lignin (N/lignin). The litter C and N concentrations were determined by dry combustion (Bremner, 2018) using a LECO CN-628 analyzer (LECO Inc., St. Joseph, MI, USA). Separate litter subsamples were digested using 1 mol/L hydrochloric acid to measure P, K, Ca, and Mg concentrations by the Inductively Coupled Plasma Optical Emission Spectrometry technique (ICP-OES, Soltanpour et al., 2018) using a Perkin Elmer Avio 200 instrument (PerkinElmer, Inc., MA, USA). Litter lignin concentration was measured using Association of Official Agricultural Chemists (AOAC) Official Method “973.18-Fiber Acid Detergent and Lignin in Animal Feed” (AOAC 1996) by the University of Florida Forage Evaluation Support

Table 1
Dominant plant species and their relative abundance in the field and in the litterbags of each wetland type.

Wetland type	Plant species	Frequency in wetlands	Relative abundance in wetlands	Relative abundance in litterbags
IM × fenced × burned	<i>Panicum hemitomon</i>	0.52	0.39	1.00
	<i>Panicum hemitomon</i>	0.61	0.6	0.78
IM × fenced × unburned	<i>Hymenachne amplexicaulis</i>	0.17	0.17	0.22
	<i>Panicum hemitomon</i>	0.40	0.14	0.42
	<i>Paspalum notatum</i>	0.19	0.11	0.33
IM × grazed × burned	<i>Juncus effusus</i>	0.48	0.08	0.24
	<i>Luziola fluitans</i>	0.60	0.23	0.41
	<i>Juncus effusus</i>	0.53	0.18	0.32
	<i>Panicum hemitomon</i>	0.45	0.15	0.27
SN × fenced × burned	<i>Panicum hemitomon</i>	0.67	0.39	0.85
	<i>Rhynchospora inundata</i>	0.32	0.07	0.15
SN × fenced × unburned	<i>Panicum hemitomon</i>	0.45	0.4	0.60
	<i>Rhynchospora inundata</i>	0.51	0.21	0.31
	<i>Amphicarpum muehlenbergianum</i>	0.23	0.06	0.09
	<i>Rhynchospora inundata</i>	0.48	0.15	0.27
SN × grazed × burned	<i>Panicum hemitomon</i>	0.33	0.14	0.25
	<i>Eleocharis vivipara</i>	0.17	0.1	0.18
	<i>Pontederia cordata</i>	0.21	0.07	0.13
	<i>Bacopa caroliniana</i>	0.28	0.07	0.13
	<i>Luziola fluitans</i>	0.20	0.03	0.05
	<i>Panicum hemitomon</i>	0.50	0.18	0.31
	<i>Pontederia cordata</i>	0.28	0.15	0.26
SN × grazed × unburned	<i>Luziola fluitans</i>	0.36	0.14	0.24
	<i>Rhynchospora inundata</i>	0.27	0.11	0.19

Laboratory. Based on the measurement of litter traits of each plant species (Table S1), the overall litter traits for the litterbags deployed to each corresponding wetland type were calculated using abundance-weighted averages (Eq. 3), which are shown in Table S2.

$$W = \sum_{i=1}^n w_i T_i \quad (3)$$

where W is the weighted average for a given trait, n is the total number of dominant plant species in litterbags for each type of wetland, w_i is the relative abundance of species i in the litterbag (Table 1), and T_i refers to the average measured trait value of species i .

Soil physiochemical properties were obtained using soil samples collected in both March and June 2019 at each sampling site in order to capture potential seasonal variations. More specifically, we measured water content, pH, organic matter content, total C and N contents, and plant-available (i.e., Mehlich-3) P, K, Ca, Mg contents. Detailed sampling and measurement protocols can be found in Guo et al. (2021). Briefly, three top 5-cm soil cores were collected and mixed at each site, then sieved through 2-mm mesh and stored within sealed plastic bags in 4 °C until further analyses. Gravimetric water content was measured using weight loss after oven drying at 105 °C for 72 h. Soil pH was measured in a 1:1 soil/distilled water solution using a pH meter (Hanna Instruments, CA, USA) (Thomas, 1996). Organic matter content was measured by the weight loss-on-ignition method (Schulte and Hopkins, 2015) by placing soil in a muffle furnace at 450 °C for 4 h. Soil total C and N contents were measured using a LECO CN-628 analyzer (LECO Inc., St. Joseph, MI, USA). Plant-available P, K, Ca and Mg were extracted using Mehlich-3 solution (Mehlich, 1984), then determined by the ICP-OES technique (Soltanpour et al., 2018) using a Perkin Elmer Avio 200 instrument (PerkinElmer, Inc., MA, USA). The soil properties across wetland treatments are found in Fig. S1.

Soil subsamples for identification of bacterial and fungal communities were also collected in March and June 2019 at each sampling site. Microbial samples across different sites from the same wetland were mixed and homogenized thoroughly to represent the community condition of the entire wetland. Soil samples for microbial analyses were immediately placed on dry ice for transportation, then stored at -80 °C until DNA extraction. We assessed the composition of bacterial and fungal communities using high-throughput sequencing of bacterial 16S ribosomal RNA (16S rRNA) and fungal ribosomal internal transcribed spacer (ITS) genes (Edgar, 2018). A detailed processing protocol can be found in (Guo et al., 2022). Briefly, genomic DNA of soil microbes was extracted using the DNeasy PowerSoil extraction kit (Qiagen, Norway). The universal primer sets of 341F (5'-CCTACGGG NGGCWGCAG-3')/806R (5'-GGACTACHVGGGTWTCTAAT-3') and ITS1F (5'-CTTGGTCATTAGAGGAAGTAA-3')/ITS4 (5'-TCCTCCGCTTATTGAT ATGC-3') were used to target the V3-V4 region of bacterial 16S rRNA and fungal ITS4 genes. A two-step polymerase chain reaction (PCR) (Chen et al., 2021) was conducted to amplify target genes. Then the verified PCR products were pooled and sequenced at the Duke Center for Genomic and Computational Biology (GCB, Durham, NC, USA) in a single library using Illumina Miseq v3 kits (paired-end 2 × 300 bp, Illumina Inc., San Diego, CA, USA). The raw sequences were deposited in the public NCBI Sequence Read Archive (SRA) database under the BioProject number PRJNA824538. The amplicon sequencing data were processed in QIIME2 (version 2020.2) and denoised using DADA2 (Callahan et al., 2016). A total of 1,735,406 quality-filtered 16S rRNA gene and 561,066 ITS gene reads were obtained. The SILVA version 132 (Quast et al., 2013) and UNITE version 8.0 (Kõljalg et al., 2013) databases were used for bacterial and fungal taxonomic identification, respectively. Operational taxonomic units (OTUs) were determined at the 99 % similarity level of the nucleotide sequences. Bacterial functional annotations were assigned according to FAPROTAX (Louca et al., 2016) databases. Alpha diversity of microbial communities was characterized by assessing richness, Shannon diversity index, and Pielou's evenness at the levels of both OTU and functional group. Details on the microbial communities across wetland treatments are shown in Figs. S2 and S3.

2.5. Statistical analyses

All statistical analyses were performed in R version 4.1.2 (R Core Team, 2021) using RStudio platform. We used linear mixed-effects models to determine individual and interactive effects of wetland management on decomposition rate k and recalcitrant fraction A , respectively (i.e., Question 1). Wetland treatments (i.e., upland land-use intensification × grazing × fire) were treated as fixed factors, while sampling site (i.e., edge, middle, and center sites) and block were treated as random factors. Assumptions of all specified model were checked by visually inspecting residual plots, and no violations were detected. We estimated p -values using Type III Sums of Squares, because our variables were non-sequential and the interactions among variables were considered. We used pairwise contrasts for post-hoc tests on any significant interactions. Linear mixed-effects models and pairwise comparisons were performed using “lmer” and “contrast” functions from *lme4* (Bates et al., 2014) and *lsmeans* (Lenth, 2016) packages, respectively.

Multiple regression and model selection were performed to rank variables related to litter traits, soil properties, and soil microbial attributes, and identify the most important factors driving in situ plant litter decomposition (i.e., Question 2). According to (Carrillo et al., 2017), our analysis included two rounds of model selection: the first round of model selection aimed to select best significant variables within each predictor category (i.e., litter traits vs. soil properties vs. microbial attributes vs. variable interactions); and the second round of model selection aimed to select the most important variables across all categories of significant predictors. All possible multiple regression models of k and A were compared based on Akaike Information Criterion (AICs), then the four best models for each type of variables were presented in Table 3. Model selection and ranking was performed using the “dredge” function from *MuMIn* package (Barton and Barton, 2015).

Based on the results from the above multiple regressions and model selection, we constructed structural equation models (SEM) to identify direct and indirect pathways through which land management affected litter decomposition (i.e., Question 3). SEM were constructed separately for k and A . Prior to SEM, hypothetical path diagrams were conceptualized (Fig. S4), and all included variables were standardized to 0–1 scale for comparing magnitude of effects across predictors. Non-significant terms were removed iteratively during model fitting until the fitness of SEM model was adequate with low Fisher's C value and non-significant chi-squared tests. SEM was performed using the “psem” function in *piecewiseSEM* package (Lefcheck, 2016).

3. Results

Overall, our results showed that land management exerted significant individual or interactive effects on in situ plant litter decomposition processes. Firstly, upland land-use intensification and cattle grazing interactively affected decomposition rate k (Table 2). Land-use intensification increased k by 31.1 % in ungrazed wetlands but decreased k by 13.0 % in grazed wetlands (Fig. 2A). Cattle grazing resulted in 15.1 % higher k -values in SN wetlands, but 23.6 % lower k -values in IM wetlands (Fig. 2A). Secondly, prescribed fire exerted individual effects on both k and recalcitrant fraction A (Table 2), specifically burned wetlands had 17.5 % lower k -values (Fig. 2B) and 40.9 % higher A -values than unburned wetlands (Fig. 2D). Finally, land-use intensification reduced A by 24.3 %, regardless of grazing or fire treatments (Table 2, Fig. 2C). Additionally, we performed similar and complementary analyses and found that no significant spatial variations in litterbag decomposition within a wetland or across blocks (Fig. 3), suggesting that individual wetland is an appropriate spatial scale for inference.

Among sets of predictive variables from each category, regressions against litter traits produced the models with the highest explanatory powers for both decomposition parameters (k : 0.551 < R^2 < 0.561; A : 0.332 < R^2 < 0.424), followed by regressions against soil properties (k : 0.201 < R^2 < 0.404; A : 0.256 < R^2 < 0.335). Measurements of soil bacterial and fungal communities did not independently generate any significant models for predicting decomposition processes (i.e., p value > 0.05) (Table 3). However,

Table 2

Linear mixed-effects model results on litterbag decomposition rate k and recalcitrant fraction A in response to management factors. (Bold values indicate significant coefficients, * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$).

Management treatment	Estimate \pm SE	F value	p value
Decomposition rate k			
Land-use intensity (IM)	0.0003 \pm 0.0002	2.321	0.131
Cattle grazing (Grazed)	0.0001 \pm 0.0002	3.245	0.075
Prescribed fire (Burned)	-0.0009 \pm 0.0002	20.512	<0.001***
IM \times Grazed	-0.0007 \pm 0.0003	22.564	<0.001***
IM \times Burned	0.0005 \pm 0.0003	2.313	0.132
Grazed \times Burned	0.0005 \pm 0.0003	2.570	0.112
IM \times Grazed \times Burned	-0.0005 \pm 0.0004	1.352	0.248
Recalcitrant fraction A			
Land-use intensity (IM)	0.0060 \pm 0.0793	4.264	0.042*
Cattle grazing (Grazed)	0.0647 \pm 0.0848	0.965	0.328
Prescribed fire (burned)	0.2412 \pm 0.0780	6.423	0.013*
IM \times Grazed	-0.0212 \pm 0.1142	1.018	0.315
IM \times Burned	-0.2526 \pm 0.1092	3.794	0.054
Grazed \times Burned	-0.1314 \pm 0.1143	0.159	0.691
IM \times Grazed \times Burned	0.2003 \pm 0.1567	1.634	0.204

certain interaction terms between microbial and litter or soil variables were significant (Table 3).

Selected best predictors from multiple regressions (Table 3) varied across categories of litter, soil, and microbial variables for k and A , indicating that factors controlling decomposition rates and recalcitrant fractions were different (Table 3). Our identified best SEM demonstrated that land

management affected decomposition processes via both direct and indirect pathways through altering litter traits, soil properties, and interactions with soil and microbial variables (Figs. 4, 5). The SEM of k (Fisher's $C = 17.0$, $p = 0.91$) explained 83 % of the variation in decomposition rates (Fig. 4). Upland land-use intensification resulted in wetland plant litter containing higher P but lower Mg, both of which were negatively associated with k (both $p < 0.001$; effect size = -0.61, -1.12). Land-use intensification decreased relative abundance of cellulolytic bacteria, which had a positive effect on k ($p < 0.05$; effect size = 0.31). Cattle grazing resulted in higher soil K, which had a positive effect on relative abundance of cellulolytic bacteria ($p < 0.001$; effect size = 0.43), and interactively affected k with cellulolytic bacteria ($p < 0.001$; effect size = -0.73). In addition to the above indirect effects, cattle grazing exerted a direct positive effect on k ($p < 0.001$; effect size = 0.13). Prescribed fire effects on decomposition k were mostly through affecting litter P content (Fig. 4).

The SEM of A (Fisher's $C = 18.0$, $p = 0.80$) explained 56 % of the variation in recalcitrant fractions (Fig. 5). Land-use intensification and prescribed fire led to a higher plant litter lignin, which was positively associated with A ($p < 0.01$; effect size = 0.36). Land-use intensification increased soil pH, making the soil pH more neutral, which promoted soil bacterial functional diversity ($p < 0.001$; effect size = 0.57). Moreover, land-use intensification increased soil water content of seasonal wetlands, which resulted in higher A ($p < 0.001$; effect size = 0.96) and was positively correlated with soil N ($p < 0.001$). Soil N and bacterial functional diversity interactively led to a lower A ($p < 0.05$; effect size = -0.64). Furthermore, land-use intensification exerted a direct negative effect on A ($p < 0.001$; effect size = -0.23).

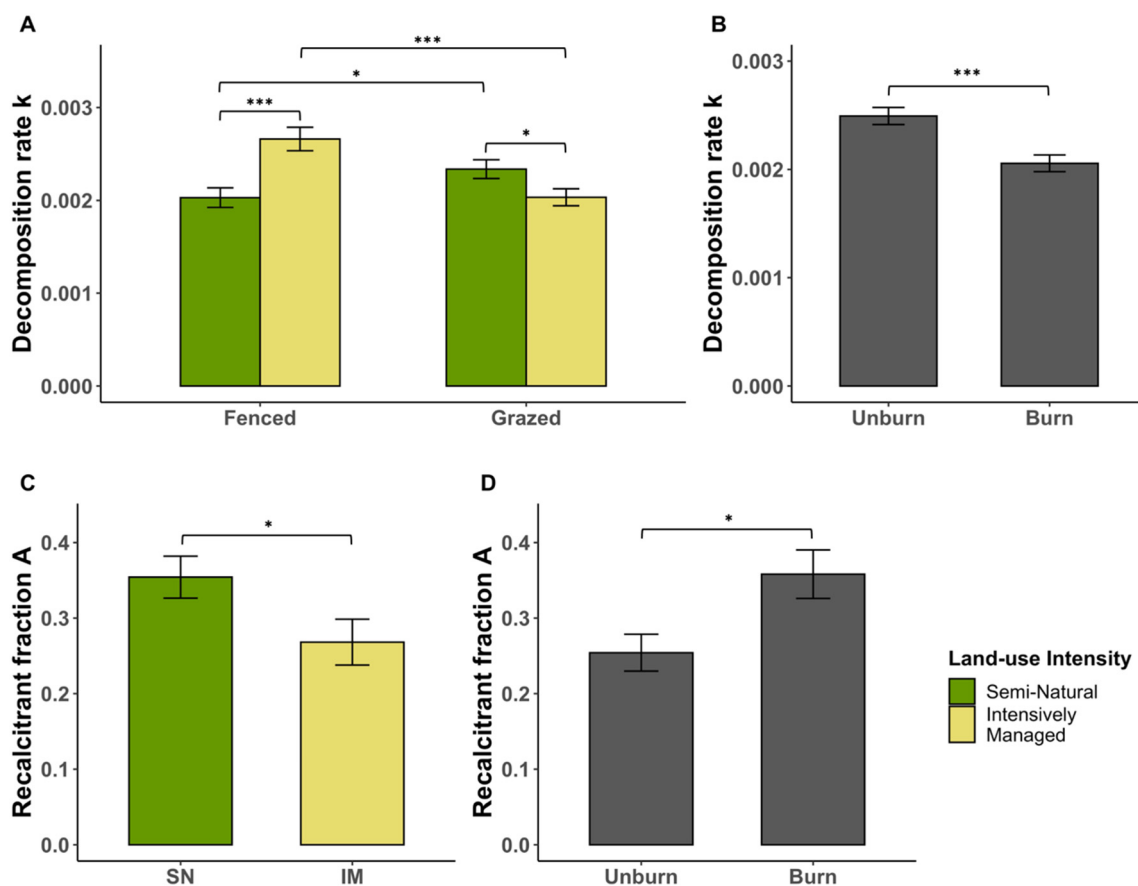


Fig. 2. Significant differences in litterbag decomposition rate k (A, B) and recalcitrant fraction A_{sym} (C, D) across wetland treatments (from embedded in intensively managed (IM) and semi-natural (SN) pastures, grazed and fenced (ungrazed) wetlands, burned and unburned wetlands). Visualizations of bar plots were simplified to illustrate differences among treatment with significant individual or interactive effects. Brackets indicate the pairwise comparison (* indicates $p < 0.05$; ** indicates $p < 0.01$; *** indicates $p < 0.001$). Error bars represent SEs, and brackets across bars indicate significant differences based on post-hoc pairwise comparisons. Full results of model fitting are shown in Table 1.

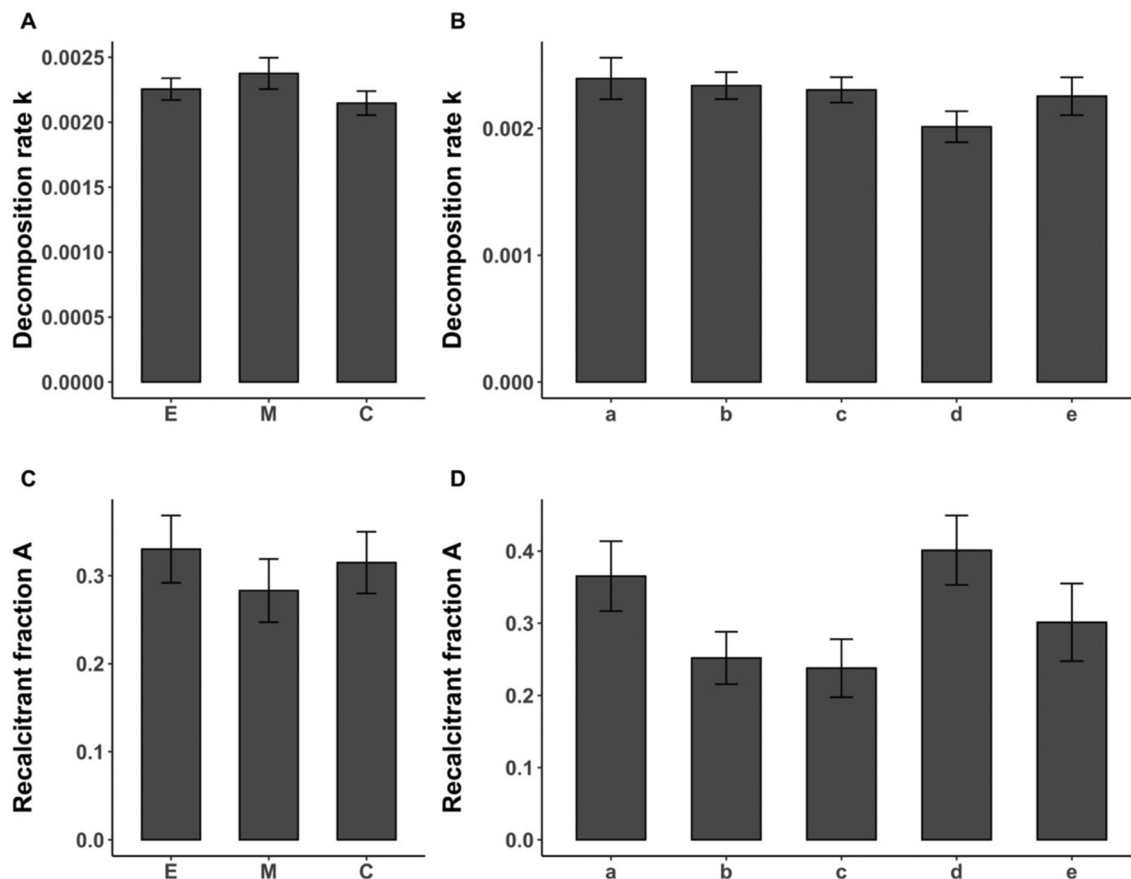


Fig. 3. Differences in litterbag decomposition rate k (A, B) and recalcitrant fraction A_{sym} (C, D) across three wetland sampling sites (Edge, Middle and Center) and five blocks (a, b, c, d, e). Error bars represent SEs.

4. Discussions

In this study, we investigated individual and interactive effects of land-use intensification, cattle grazing, and prescribed fire on processes of in situ plant litter decomposition in subtropical seasonal wetlands. We also evaluated both direct and indirect pathways of land management effects through changes in wetland biotic and abiotic conditions. Overall, our results showed that wetland litter decomposition rates and recalcitrant fractions were either individually or interactively affected by land management. While certain land management effects were direct, most significant and pronounced effects were manifested indirectly through alterations in litter traits, soil properties, and microbial attributes.

4.1. Land management affected litter decomposition by altering litter traits and associated interactions

Consistent with prior research (Ge et al., 2013; Krishna and Mohan, 2017), litter traits were the most dominant factor influencing the rates and completeness of decomposition processes in subtropical wetlands, as compared to other biotic and abiotic factors (Table 3). It was also the major pathways through which wetland management manifested their effects on litter decomposition (Figs. 4, 5). For example, land-use intensification and cattle grazing interactively affected k (Fig. 2), primarily because they exerted opposite effects on litter Mg concentration, which was one of the strongest predictors of k -values (Fig. 4). Mg is an essential element in plant cells and involved in building conformational stabilization of macromolecules, including cell walls, cell membranes, proteins, and nucleic acids (Guo et al., 2016). Therefore, higher litter Mg concentrations indicate higher capability of stabilizing conformational structures in plant tissues that could later slow down its decomposition, which explains the negative relationship between litter Mg and k -values (Table 3, Yue et al. (2021)).

Negative effects of land-use intensification on litter Mg concentration are likely due to fertilization with K, which can cause plant Mg deficiency (Guo et al., 2016), because K ions are antagonists for the absorptions of Mg ions in plants (Xie et al., 2021). Such antagonistic effects of K ions on Mg ions uptake by plants also explained why land-use intensification increased contents of both plant-available K and Mg in wetland soils (Table S4), but only promoted litter K concentration and reversely decreased litter Mg concentration (Table S3). Positive effects of cattle grazing on litter Mg concentration are likely driven by grazing-induced shifts in wetland plant community composition. Indeed, Boughton et al. (2016) found that, in these wetlands, cattle grazing promoted growth of short graminoids and forbs (e.g., *Eleocharis vivipara*, *Luziola fluitans*, and *Bacopa caroliniana*), which were shown to have higher content of Mg than other common grasses and forbs in these wetlands (Table S1) and also in grasslands of other geographic regions (Reiné et al., 2020).

Prescribed fire exhibited a suppression effect on litter decomposition by promoting growth of plants with high concentrations of P and lignin, which was negatively associated with k -values (i.e., reduced decomposition rates) and positively associated with A -values (i.e., increased fraction of recalcitrant litter and thus decreased completeness of decomposition), respectively (Figs. 4, 5). Prescribed fire has been found to increase plant tissue P concentration and thus alter litter quality in multiple other studies (Butler et al., 2018; Cui et al., 2010; Wang et al., 2014), because burning could increase soil labile P content (as also shown in our data; Table S4) and thereby facilitating plant P absorption (Butler et al., 2018). Unlike many studies measuring responses of plant nutrient status to fire disturbances, very few studies examined responses of plant lignin biosynthesis to periodical burning (Xie et al., 2018). Our results showed that vegetation in burned wetlands contained more lignin, possibly because lignin in cell wall provides structural support and functions as physical barrier. Thus plants with high level of lignin tend to be favored evolutionarily to build resistance and

tolerance of heat stress under repeated burns (Liu et al., 2018). As expected, higher litter lignin content led to higher *A*-values, since lignin is a well-known contributor to the recalcitrance of organic matter residues (Chapin et al., 2002; Swift et al., 1979). However, we surprisingly found higher litter P concentration resulted in slower decomposition rates, which is contradictory to many other studies, such as Wardle et al. (2002) and Zheng et al. (2017). Nevertheless, results from Chen et al. (2013) and van Huysen et al. (2016) in naturally nutrient-rich systems were consistent with our observations. We suspect that these ‘seemingly’ divergent results might arise from the complex interaction of litter traits with soil properties or microbial

decomposers on litter decomposition (Carrillo et al., 2017; Zhan et al., 2021), as also shown in our results (Table 3). Due to the severe P legacy in Florida's agricultural soils (Sharpley et al., 2013), wetlands embedded in IM pastures were not P-limited systems (Ho et al., 2018). Thus, litter with increased P concentration would not necessarily stimulate decomposers' activities, and could even indicate that P-enriched soils likely already satisfied the microbial P demands and then inhibit the microbial P mining from fresh litter (Chen et al., 2013). To further improve the predictive power of litter traits, increasing research has revealed that plant litter physical traits (e.g., specific leaf area) (Makkonen et al., 2013) and

Table 3

Summary of best four multiple regression models of litterbag decomposition rate *k* and recalcitrant fraction *A* against grouped sets of potential predictors. The selected models were presented in the order of increasing AIC. Direction of effects were indicated with plus or minus symbol (* indicates $p < 0.05$; ** indicates $p < 0.01$; *** indicates $p < 0.001$).

Model No.	Decomposition rate <i>k</i>				Recalcitrant fraction <i>A</i>			
	1	2	3	4	1	2	3	4
Model term	R ² =0.551	R ² =0.561	R ² =0.561	R ² =0.556	R ² =0.408	R ² =0.424	R ² =0.332	R ² =0.410
Litter traits								
C/N ratio								0.042
N			0.0002					
P	-0.0013***	-0.0011**	-0.0011**	-0.0014***				
K				0.0002		-0.076		
Mg	-0.0021***	-0.0018***	-0.0018***	-0.0021***			0.233**	
Ca					0.230**	0.261***		0.214**
Lignin					0.270**	0.252**		0.249**
N/lignin ratio		0.0002						
	R ² =0.404	R ² =0.310	R ² =0.201	R ² =0.351	R ² =0.317	R ² =0.256	R ² =0.335	R ² =0.278
Soil properties								
C/N ratio				-0.0003				
TN					-0.284*		-0.361*	
Mehlich-3 P	-0.0013***	-0.0009**		-0.0011**				
Mehlich-3 K			-0.0007*				0.128	
Mehlich-3 Ca	0.0006.			0.0006.		-0.226*		-0.181
pH					-0.275**		-0.266**	-0.118
Water content					0.562**	0.372**	0.523**	0.343**
	R ² =0.060	R ² =0.040	R ² =0.036	R ² =0.036	R ² =0.133	R ² =0.110	R ² =0.142	R ² =0.089
Microbial diversity	All models below were insignificant				All models below were insignificant			
Fungal richness	0.0005							-0.089
Fungal Shannon		0.0005						
Fungal evenness			0.0004			-0.111		
Bacterial richness				-0.0005				
Bacterial functional Shannon index					-0.169		-0.318	
Bacterial functional richness							0.201	
	R ² =0.051	R ² =0.012	R ² =0.011	R ² =0.011	R ² =0.132	R ² =0.178	R ² =0.110	R ² =0.109
Microbial group relative abundance	All models below were insignificant				All models below were insignificant			
Cellulolytic bacteria	-0.0004					0.162		
Photoheterotrophs		0.0003					-0.184	
Chemoheterotrophs			-0.0003		0.209			
Phototrophs				0.0003				-0.185
	R ² =0.253	R ² =0.224	R ² =0.208	R ² =0.399	R ² =0.490	R ² =0.464	R ² =0.489	R ² =0.413
Interactions between litter traits and microbial features								
Litter P: bacterial richness				-0.0014**				
Litter Mg: cellulolytic bacteria			-0.0009**					
Litter Mg: chemoheterotrophs	-0.0011**			-0.0019***				
Litter Mg: bacterial richness		-0.0010**						
Litter Ca: bacterial functional richness					0.423***		0.382.	
Litter Ca: photoheterotrophs						0.649**	0.093	
Litter Ca: cellulolytic bacteria								0.224**
Lignin: chemoheterotrophs					0.405***	0.433***	0.415***	0.374***

	R ² =0.157	R ² =0.170	R ² =0.211	R ² =0.228	R ² =0.299	R ² =0.345	R ² =0.350	R ² =0.280
Interactions between soil properties and microbial features								
Soil K: bacterial richness	-0.0011*							
Soil P: bacterial richness		-0.0011*						
Soil K: cellulolytic bacteria			-0.0008*					
Soil P: cellulolytic bacteria				-0.0009*				
Soil N: bacterial functional richness					-0.399*	-0.349*		
Soil N: bacterial Shannon index							-0.351*	-0.346*
Soil water: cellulolytic bacteria					0.779			0.653***
Soil water: chemoheterotrophs						0.916***	0.824***	
Soil pH: chemoheterotrophs						-0.321	-0.320	
	R ² =0.720	R ² =0.714	R ² =0.720	R ² =0.726	R ² =0.512	R ² =0.538	R ² =0.504	R ² =0.523
Combined significant variables								
Litter P	-0.0011***	-0.0011***	-0.0011***	-0.0011***				
Litter Mg	-0.0019***	-0.0016***	-0.0019***	-0.0016***				
Litter Ca					0.197**	0.183**	0.203**	0.175*
Lignin					0.246**	0.236**	0.249**	0.242**
Litter Mg: bacterial richness		-0.0004						
Litter Mg: chemoheterotrophs				-0.0004				
Soil K	-0.0007***	-0.0007***	-0.0005	-0.0007***				
Soil N					-0.234*	-0.178*	-0.012	-0.096
Soil pH					-0.187*	-0.148		-0.144
Soil water					0.527***		0.524***	0.553***
Soil K: cellulolytic bacteria			-0.0002					
Soil N: bacterial functional Shannon							-0.420	-0.301
Soil water: chemoheterotrophs						0.804***		

heavy metal concentrations, such as Mn (Berg, 2014) could affect litter decomposition, which would warrant more attention in future studies.

4.2. Land management affected litter decomposition by altering soil properties and associated interactions

Soil properties were the second most important predictors in explaining variations of decomposition processes (Table 3). Specifically, we found that

soil plant-available P and K content negatively affected *k*-values, whereas soil TN and pH negatively but water content positively regulated *A*-values (Table 3). Our SEM results further showed that certain effects of soil properties were manifested through their interactions with other soil or microbial variables (Figs. 4, 5). In wetlands, upland land-use intensification and wetland on-site disturbances significantly altered soil physiochemical properties, particularly soil plant-available P and K content, pH, and water content (Table S3). These results suggested that changes in soil properties

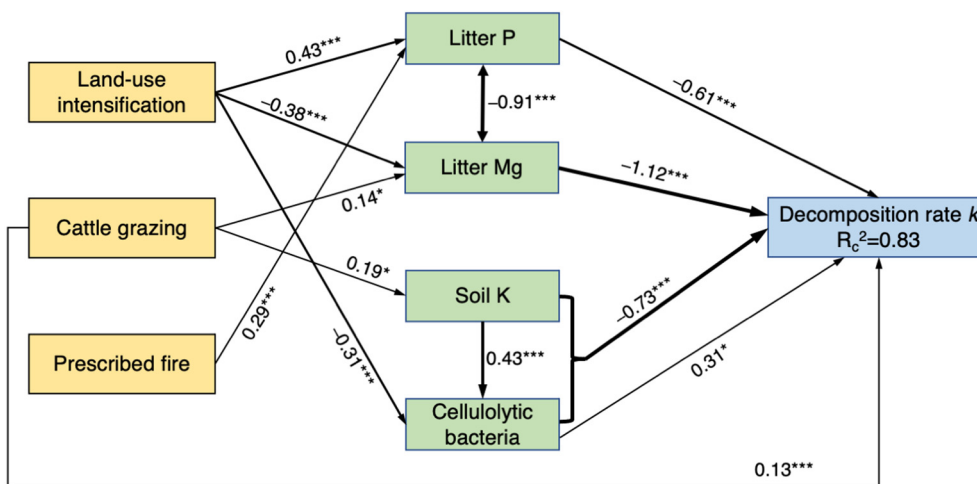


Fig. 4. Structural equation modeling results to illustrate direct and indirect pathways of land management effects on decomposition rate *k*. Black arrows represented each significant path (* indicates *p* < 0.05; ** indicates *p* < 0.01; *** indicates *p* < 0.001), with brackets indicating interaction effects and double arrowed lines indicating correlation without establishment of causality. Nonsignificant treatment effects or relationships (*p* > 0.05) are omitted in the figure for clarity. The thickness of arrows represents the level of standardized effect sizes.

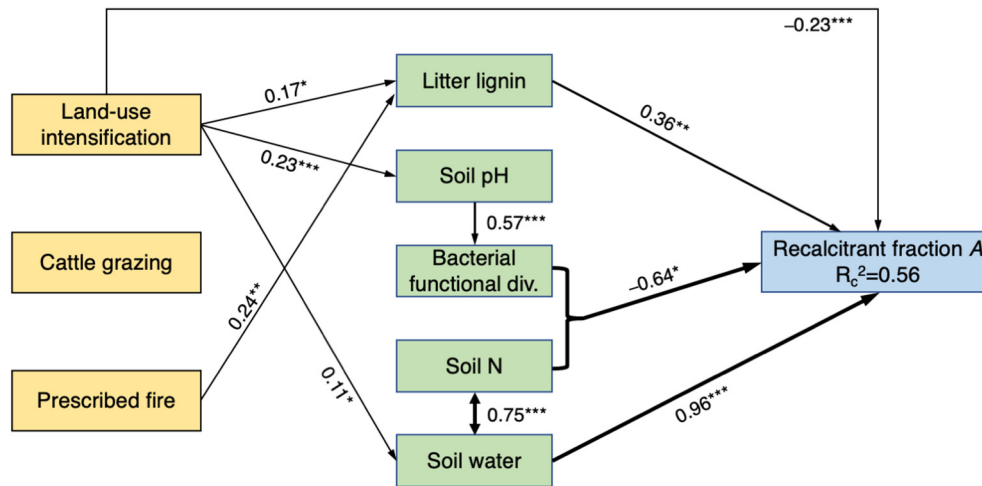


Fig. 5. Structural equation modeling results to illustrate direct and indirect pathways of land management effects on recalcitrant fraction A. Black arrows represented each significant path (* indicates $p < 0.05$; ** indicates $p < 0.01$; *** indicates $p < 0.001$), with brackets indicating interaction effects and double arrowed lines indicating correlation without establishment of causality. Nonsignificant treatment effects or relationships ($p > 0.05$) are omitted in the figure for clarity. The thickness of arrows represents the level of standardized effect sizes.

induced by land management are the additional indirect pathways through which land management affects wetland litter decomposition.

Our results regarding the overall inhibitory effect of nutrient enrichment (due to land-use intensification or grazing) on decomposition rates (Table 3; Fig. 4) are consistent with findings from the teabag experiment in these same wetlands (Guo et al., 2021) and other decomposition studies (Chen et al., 2013; Tie et al., 2022; Zheng et al., 2017). Further, our results also aligns with the teabag observations (Guo et al., 2021), which found that higher soil nutrient levels only slowed down decomposition rate (i.e., lower k), which represents the initial stage of decomposition process that are mainly controlled by labile organic matter. However, nutrient addition facilitated the decomposition completeness by reducing recalcitrant fractions (i.e., lower A) (Table 3), which represents the relative long-term decomposition process that is mainly controlled by complex macromolecules (Chapin et al., 2002). Collectively, these results suggest that nutrient enrichment induced by land management could exhibit mixed effects on different stages in decomposition depending upon molecular composition in plant litter.

Moreover, we found more neutral pH environment that were due to upland intensification (Fig. 5) and less soil water content were conducive to a more complete decomposition of recalcitrant organic compounds in wetlands, which is consistent with studies in wetland ecosystems (Middleton, 2020; Mueller et al., 2018; Petraglia et al., 2019; Wright et al., 2013). The suppression effect of high soil water content on decomposition completeness in wetlands could be caused either by water's high heat capacity that results in low temperature in hot climates and small temperature variation during the entire incubation (Petraglia et al., 2019) or by longer inundation and subsequently low oxygen availability, thus reducing the activities of aerobic decomposers (Mueller et al., 2018). A promotion effect on decomposition completeness at neutral soil pH was likely because that a pH-neutral environment favors the microbial diversity and activities (Yang et al., 2021). Our microbial data collected from these wetlands confirmed this proposed mechanism by showing that more neutral soil pH was significantly correlated with higher bacterial functional diversity (Guo et al., 2022), thus contributing to a more thorough litter decomposition that was indicated by reduced recalcitrant fractions.

Our results showed that dominant drivers for each decomposition stage (i.e., k vs. A) were different (Table 3), suggesting varying effects of external factors on litter molecular components. For example, we only observed significant effects of soil water content and pH on A -values instead of k -values (Table 3). It was probably because, compared to labile organic material, the breakdown of aged and recalcitrant organic matter was more dependent on the actions of heterotrophic microorganisms and thus more sensitive to the

absence of oxygen and level of pH (Kristensen et al., 1995; Mueller et al., 2018). Such results allude to the complexity and nuances in the decomposition processes, and highlight the importance of linking decomposition of specific litter components to their corresponding predictors.

Finally, our results showed relatively weak or no individual effects of community-level microbial attributes on wetland litter decomposition. Rather, their effects manifested through interactions with soil properties (Table 3; Figs. 4, 5). Specifically, soil K content and relative abundance of cellulolytic bacteria interacted to negatively affected decomposition rates, while cellulolytic bacteria independently showed a positive effect on k -values (Fig. 4). These results suggest that higher relative abundance of cellulolytic bacteria could indicate more active cellulose-degrading activities, but only when the level of soil K is low. It is expected that more cellulolytic bacteria could contribute to faster early-stage decomposition rates because cellulose is considered a labile litter component (Stagg et al., 2018). We suspected that soil K enrichment might exhibit similar inhibitory effects as soil N addition on the decomposition of labile organic compounds (Chen et al., 2019). However, whether and how that soil K inhibited the enzyme expressions of cellulolytic bacteria would need further molecular studies to confirm. In addition, soil N and bacterial functional diversity showed a synergic effect on reducing litter recalcitrant fractions (Fig. 5), suggesting functional traits of soil microbes and nutrient status of soil collectively determined the level of organic matter depletion.

4.3. Management implications

Our research provides a scientific basis for the sustainable management of tropical and subtropical wetlands that are experiencing increasing pressures from land management and anthropogenic environmental changes. Our results indicate that human-induced disturbances in wetlands, particularly upland agricultural intensification, could accelerate litter decomposition rates and promote long-term litter decomposition completeness (Fig. 2), which would consequently go against the formation of stable organic matter and decrease the capacity of seasonal wetlands to retain nutrients and sequester carbon. However, there are nuanced interactive effects from these management practices, in which cattle grazing could help buffer against the stimulating effect of land-use intensification on decomposition rates and thus avoid nutrient release pulses. Further, periodic prescribed fire showed profound inhibitory effects on wetland decomposition, suggesting that fire suppression could potentially further accelerate decomposition. Therefore, to better maintain regulating and supporting services provided by wetlands (e.g., water purification, nutrient retention, and carbon sequestration), it is

important to reduce anthropogenic disturbances from agricultural activities to wetland ecosystems (Guo et al., 2021). For instance, we should consider avoiding agricultural intensification in low-lying grasslands where more seasonal wetlands are concentrated and avoiding fire suppression by adopting prescribed fire practices for wetlands embedded in tropical/subtropical agroecosystems.

Moreover, we found that cattle grazing exhibited weaker effects on litter decomposition than upland land-use intensification and prescribed fire (Table 2). This is probably because cattle grazing could exert multiple opposite effects on litter decomposition that offset each other. For example, our SEM showed that cattle grazing increased litter Mg and K content which in turn decreased litter decomposition rate. However, it also directly increased litter decomposition rate. This direct effect could be due to grazing also reduced vegetation aboveground biomass and increased the abundance of short graminoids and forbs (Boughton et al., 2016), which might increase solar radiation exposure to the ground litter, result in higher soil temperature and greater photodegradation of recalcitrant litter fractions that consequently facilitates its decomposition (Austin et al., 2016). Therefore, it may not be necessary to remove cattle from wetlands in agroecosystems, if grazing is properly managed, at least for the purpose of maintaining natural decomposition processes.

5. Conclusions

Our research demonstrates that agricultural land use and management interact to substantially affect plant litter decomposition in wetlands either directly, or indirectly through altering litter traits, soil properties, and soil microbial communities. Specifically, we found: (1) land-use intensification increased k in ungrazed wetlands and decreased k in grazed wetlands, but consistently reduced A regardless of grazing or fire treatment; (2) prescribed fire individually suppressed litter decomposition by reducing k and increasing A ; (3) grazing effects on k depended on land-use intensification and impacts of grazing were weaker than intensification or fire; (4) litter traits were the strongest predictors of in situ wetland plant litter decomposition as compared to soil and microbial properties, and the best predictors for k and A varied; and (5) microbial traits only exhibited interactive effects with litter and soil variables to affect wetland litter decomposition. Our results highlight the need to avoid fire suppression and agricultural intensification in areas surrounding natural wetlands to maintain natural decomposition processes and in turn provision of regulating ecosystem services. Our results on wetland litter decomposition responses to land management fill empirical data gaps in earth system models for predicting effects of interactive global environmental change drivers on wetland functions and services in humid tropical/subtropical regions, relevant for sustainable agroecosystem management to conserve wetland functions and services given increasing anthropogenic environmental changes.

CRedit authorship contribution statement

Yuxi Guo: Conceptualization, Investigation, Methodology, Formal analysis, Data curation, Writing – original draft. **Elizabeth H. Boughton:** Conceptualization, Supervision, Writing – review & editing. **Hui-Ling Liao:** Methodology, Funding acquisition, Writing – review & editing. **Gregory Sonnier:** Methodology, Writing – review & editing. **Jiangxiao Qiu:** Conceptualization, Funding acquisition, Supervision, Methodology, Writing – review & editing.

Data availability

Data will be made available on request.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.scitotenv.2022.158789>.

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