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Dissolved organic matter concentration, molecular composition, and functional groups in contrasting management practices of peatlands

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Abstract

About 91,300 ha of peatlands has been rewetted in western Europe since the mid-1990s. Still, it is unknown how long-term rewetting alters the dissolved organic matter (DOM) concentration, molecular composition, and functional groups. We examined these DOM characteristics in three peatland types subjected to 47- to 231-yr drainage and 18- to 24-yr rewetting to address this knowledge gap. Cold water-extractable DOM was characterized by pyrolysis field ionization mass spectrometry (Py-FIMS) and X-ray absorption near-edge structure (XANES) spectroscopy. The dissolved organic carbon (DOC) concentration in the rewetted forest peatland was 2.7 times higher than in the drained forest peatland. However, rewetting decreased the DOC concentrations by 1.5 and 4 times in the coastal peatland and percolation mire, respectively, compared with their respective drained peatlands at the topsoil horizons. The Py-FIMS analysis revealed that all nine DOM compound classes' relative abundances differed between the rewetted and drained forest peatland with the lower relative abundances of the labile DOM compound classes in the rewetted forest peatlands. However, most DOM compound classes' relative abundances were similar between the rewetted and drained coastal peatlands and percolation mires. The XANES also revealed nine carbon and seven nitrogen functional groups with no apparent differences between the two contrasting management practices. The influence of drainage and rewetting on DOC concentration and molecular composition depends on peatland type, drainage period, rewetting intensity, and peat degradation status that should be considered in future research for understanding DOM transformation and transportation from degraded and restored peatland ecosystems.

Abbreviations: CHYDR, carbohydrates; C_{org}, soil organic carbon; DIC, dissolved inorganic carbon; DOC, dissolved organic carbon; DOM, dissolved organic matter; FATTY, Free fatty acids C_{16} to C_{34}; GHG, greenhouse gas; LDIM, lignin dimers; NMR, nuclear magnetic resonance; PEPTI, amides (amino acids, peptides, and amino sugars); PHLM, phenols + lignin monomers; Py-FIMS, pyrolysis field ionization mass spectrometry; SOM, soil organic matter; SUBER, suberin; TDN, total dissolved nitrogen; TII, total ion intensity; UV-Vis, ultraviolet-visible spectroscopy; VM, volatile matter; XANES, X-ray absorption near-edge structure.

1 | INTRODUCTION

Draining peatlands have enhanced peat decomposition and substantial carbon (C) losses in the form of greenhouse gas (GHG) emissions and dissolved organic matter (DOM) (Frank et al., 2014). These two processes are closely interlinked as GHG-producing microorganisms feed on DOM rather than solid peat substrate, and DOM practically is a source of GHG emissions from peatlands (AminiTabrizi et al., 2020; Sjögersten et al., 2016). Furthermore, the transfer of DOM from terrestrial to a freshwater ecosystem can impair water quality by surface water eutrophication and biodiversity loss (Emsens et al., 2016). Dissolved organic matter production due to peat degradation and transfers to a freshwater ecosystem can affect more than 50% of the world's terrestrial and freshwater ecosystems (Cris et al., 2014). Browning of freshwaters by DOM is typical water quality deterioration when peatlands with high soil organic matter (SOM) are subjected to drainage (Kritzberg et al., 2020). Water quality deterioration can cause a shortage of drinking water and increase water treatment costs since more than 4 trillion m^3 of freshwaters are used for different purposes per annum globally (Boretti & Rosa, 2019). Freshwater contaminated by DOM also negatively influences freshwater ecosystems by reducing light and vertical heat transfer that challenges organisms' survival and interaction (Solomon et al., 2015; van Dorst et al., 2019). Furthermore, dissolved organic C (DOC) influences soil acidity, microbial activity, nutrient availability, and mineral surface chemistry (Moore & Dalva, 2001; Philippe & Schaumann, 2014). Therefore, the DOM quality substantially affects the environmental consequences of peatland management, and its better understanding can support the sustainable use of peatlands.

Peatlands cover 505,000 km^2 globally; 13% of the world's peatland area is located in Europe (Tanneberger et al., 2017), and more than 50% of the original peatlands have been lost in this continent, with the most significant losses in the past 75 yr (Andersen et al., 2016; Krüger et al., 2015). The EU-LIFE nature program alone invested 80 projects to restore about 91,300 ha of degraded peatland habitats in western European countries. Restoration has been undertaken at 71 peatland sites in Germany between 1993 and 2015 (Andersen et al., 2016). Although these peatlands have been rewetted in the 1990s, their initial drainage times, historical land uses, degradation status, and peat characteristics were quite different at the time of rewettings. For instance, the three predominant fen peatlands, like forest peatland, coastal peatland, and percolation mire in northeastern Germany, have been drained from 47 to 231 yr before rewetting in the 1990s (Weil et al., 2020). The biogeochemical processes of these different peatland types with considerable variations in peat characteristics and historical land uses may respond differently to long-term drainage and rewetting. However, different fen peatland types were not considered in most previous studies investigating DOM production, quality, and dynamics in long-term drainage and rewetting peatlands.

Core Ideas

- Dissolved organic matter (DOM) concentrations and molecular compositions depend on degradation status in drained peatlands.
- Rewetting effects on DOM concentrations and molecular compositions differ with peatland types.
- Pyrolysis field ionization mass spectrometry reveals management effects on DOM molecular compositions compared with X-ray absorption near-edge structure.

A previous study reported that rewetting highly degraded peatlands for about 1 yr substantially increased DOC concentrations compared with rewetting slightly degraded peatlands (Zak & Gelbrecht, 2007). Another study also reported restoration of degraded peatlands after 10 yr increased DOC concentrations compared with the pristine peatlands and drained peatlands (Strack et al., 2015). However, other studies reported lower DOC concentration in the rewetted than in the drained peatlands (Höll et al., 2009) and lower DOC concentrations in degraded peatlands than in the intact peat (Kalbitz & Geyer, 2002). Similar to the DOC concentration, the results of a few studies investigating the influence of rewetting peatlands on the total dissolved nitrogen (TDN) concentration reported disagreeing results. For example, a microcosm experiment conducted at 25 °C using coastal peatlands collected from the southern United States showed that the TDN concentration increased in drained peatlands compared with the rewetted peatlands (Wang et al., 2016). No differences in TDN concentration between degraded and intact peatland have been reported as well (Kalbitz & Geyer, 2002). The contrasting results of rewetting peatlands' effect on DOC and TDN concentrations could attribute to variations in peatland types, pedogenesis, peat degradation status, peat-forming plants, drainage history, and rewetting intensity. In addition, DOM transformation and translocation pathways, hydrologic connectivity, and wet–dry cycles can influence DOC concentration (Aiken et al., 2011).

The influence of DOM on ecosystem functions depends on its concentration, molecular composition, and functional groups because DOM is the largest source of energy and nutrients to terrestrial and freshwater ecosystems (Wymore et al., 2016). In addition, DOM comprises a mixture of many molecular compositions and functional groups that govern its bioavailability, stability, and reactivity (Voss et al., 2020). Overall, pollution of freshwater by DOM can negatively influence its ecosystem services such as drinking water

production, recreational values, biodiversity, and biogeochemical processes (Kritzberg et al., 2020). Furthermore, the molecular composition and functional groups also influence DOM fate in terrestrial and aquatic ecosystems since DOM degradation, transportation, and adsorption to soils and sediments depend on these DOM characteristics (Aiken et al., 2011; Bolan et al., 2011; Xenopoulos et al., 2021). Thus, understanding the DOM molecular composition and functional groups could help predict the fate of DOM in terrestrial and aquatic ecosystems.

Researchers used scanning electron microscopy, transmission electron microscopy, ultraviolet-visible spectroscopy (UV-Vis) spectra, and excitation–emission matrix spectra to determine fulvic, humic, and protein-like components in the DOM (Bu et al., 2019). Similarly, electrospray ionization Fourier transform ion cyclotron resonance mass spectrometry was used to identify condensed aromatic and phenol-type compounds (D'Andrilli et al., 2010). The use of UV-Vis, excitation–emission matrix, and nuclear magnetic resonance (NMR) mainly disclosed aromatic and carbohydrate components in northern peatlands (Tfaily et al., 2013). However, DOM may comprise a considerable number of different organic molecules as revealed by pyrolysis field ionization mass spectrometry (Py-FIMS) (Leinweber et al., 2001) and liquid chromatography–mass spectrometry (Hawkes et al., 2018). In addition, various C and nitrogen (N) functional groups in soils have been identified successfully by X-ray absorption near-edge structure (XANES) (Gillespie et al., 2015; Leinweber et al., 2010). Some previous studies used advanced spectroscopic analytical methods such as NMR, XANES, Py-FIMS, UV-Vis, and Fourier-transform infrared spectroscopy to characterize the DOM molecular compositions of peatlands (Dodla et al., 2012; Kalbitz et al., 2003; Leinweber et al., 2001). Although using these advanced analytical techniques enhanced our insights into DOM's molecular chemical compositions and functional groups, most previous studies primarily focused on the short-term impact of drying–rewetting cycles on DOM molecular compositions and biodegradability. However, the influence of short-term and long-term rewetting on DOM molecular chemical composition could be different (Höll et al., 2009). Thus, disclosing the long-term drainage and rewetting effects on the DOM molecular chemical and functional groups in forest peatland, coastal peatland, and percolation mire can provide vital information for monitoring biogeochemical processes during the restoration of degraded peatlands.

The study aimed to examine the DOC concentration, DOM molecular composition, C, and N functional groups in long-term drained (47–231 yr) and rewetted (18–24 yr) forest peatlands, coastal peatlands, and percolation mires using the Py-FIMS and XANES analyses. We hypothesized the responses of DOC concentration and DOM molecular compositions to long-term drainage and rewetting depend on management history, degradation status, rewetting intensity, and peatland types. We assume that these could be the primary reasons the previous studies reported contrasting results on the influence of drainage and rewetting peatlands on DOC concentration and molecular chemical composition. Such opposing research results could have negative policy implications on restoring drained and degraded peatlands for mitigating climate change and other ecosystem services.

2 | MATERIALS AND METHODS

2.1 | Site description and soil sampling

The soil samples were collected from the drained and rewetted peatlands of the percolation mires, forests, and coastal peatlands in Mecklenburg–West Pomerania, northeastern Germany. The forest peatlands have been drained since 1786, the coastal peatland has been drained since 1850, and the percolation mire since the 1970s (Weil et al., 2020). Parts of the drained coastal peatland, forest peatland, and percolation mire started rewetting in 1993, 1998, and 1999, respectively, by rising water tables passively; that means the previous drainage ditches have been closed. Thus, the forest peatland has been drained for 231 yr, coastal peatland for 167 yr, and the percolation mire for 47 yr until the date of soil sampling for the present study in October 2017. At the time of soil sampling, the rewetted forest peatland and percolation mire were rewetted for 18 and 19 yr, respectively, and the coastal peatland was rewetted for 24 yr. In addition to rewetting by rising water tables, the rewetted percolation mire and rewetted coastal peatland have been flooded occasionally by the nearby Trebel River and brackish Baltic Sea, respectively. As a result, during short dry summers (June to August), the water tables of most sites have been lowered. However, the water table of the rewetted percolation mire remained relatively stable between −10 to 10 cm soil depths throughout the year (Ahmad et al., 2020).

At soil sampling, the drained coastal peatland and percolation mire were used for grazing and pasture production, respectively, whereas the drained and rewetted forest peatlands were used for logging. The rewetted coastal peatland and rewetted percolation mire were under nature conservations. We present the depth of the water table, peat thickness, georeferences of sampling sites, altitude, predominant plant species, mean annual temperature, and rainfall in the supporting material (Supplemental Table S1). The soil order of all study sites is Histosols and the great groups of the soils tentatively classified as Haplohemists for the rewetted forest peatland, drained and rewetted percolation mires; and Haplosaprists for the drained forest peatland, drained and rewetted coastal peatlands according to the USDA Soil Taxonomy (Soil Survey Staff, 2014).

We collected the soil samples from six sites recently established to study matter turnover processes in the drained and rewetted peatlands in northeastern Germany (http://www.wetscapes.uni-rostock.de). The six sites comprised drained and rewetted forest peatlands, coastal peatlands, and percolation mires. The recently established area of each of the six sites is (35 × 6.5 m (227.5 m^2) that was well illustrated and described by Jurasinski et al. (2020). We collected three replicate samples every 12-m interval from each site in October 2017. Soil samples were obtained from four horizons at all sites except for the rewetted coastal peatland, in which three soil horizons were sampled. The depths of soil sampling were 0–10, 10–20, 20–40, and 40–60 cm for the drained forest, coastal peatland, and drained percolation mire and were 0–20, 20–40, 40–60, and 60–80 cm for the rewetted forest peatland and rewetted percolation mire, whereas 0–10, 10–20, and 20–30 cm for the rewetted coastal peatland. Overall, we collected 69 soil samples from three drained peatlands and three rewetted peatlands. We transported the soil samples to the laboratory in a cooling box and stored them in a freezer at −20 ˚C immediately until required for different analyses.

2.2 | Dissolved organic matter extraction and chemical analyses

The DOM was extracted by cold water following the procedure described by Bijay-Singh et al. (1988). First, we extracted field moist soil samples with ultrapure water in 50-ml centrifuge tubes in a 1:6 ratio of a dry soil sample (g) to water (ml). Before starting the water extraction, we determined the soil water content of the subsample of each moist soil sample by oven drying at 105 °C for 24-h. Then, we considered the samples' soil water content to adjust to the 1:6 soil/water ratio during the DOM extraction for each soil sample. Thus, the amount of water applied for the DOM extraction depended on the soil samples' initial water content. After shaking for 30 min, they were filtered through 1.0-µm glass fiber filters (GF6, Carl Roth GmbH + Co. KG). We used a 1-µm pore size filter instead of the 0.45-um filter in the present study to include mobile colloidal matter up to 1 µm because the recent studies indicate that that the 0.45-filter pore size underestimates the DOC concentration transferred from terrestrial to freshwater ecosystems (Yan et al., 2016; Yan et al., 2018). The filtered DOM samples were used to measure the DOC and TDN concentrations, molecular chemical compositions by the Py-FIMS, and DOM functional groups by the XANES analyses. The DOC and TDN concentrations were determined by the DIMATOC 2000 (Dimatec Analysentechnik) immediately, while a portion of the DOM samples was stored at −20 ˚C until required for the Py-FIMS and XANES analyses.

2.3 | Spectrometric and spectroscopic analyses

The procedures used to characterize solid SOM compound classes by the Py-FIMS (Negassa et al., 2019) were also adopted to describe the DOM molecular compositions at the Mass Spectrometry Laboratory of Soil Science Department of the University of Rostock, Rostock, Germany. Briefly, we used about 0.20 mg of the lyophilized DOM for the Py-FIMS analysis. These samples were thermally degraded by pyrolysis in the ion source with 4.7-kV emitter and −5.5-kV counter electrode of a double-focusing Finnigan MAT 95 (Finnigan MAT). The samples were heated in a 10^{-4} Pa vacuum from 50 to 650 ˚C increments to complete pyrolysis within 15 min. We recorded mass spectra by repetitive magnetic scans in the mass to charge (*m/z*) ranging from 15 to 900 *m/z*. The emitter was flash-heated between the magnetic scans to avoid residues of pyrolysis products. We adopted the procedures recommended for the data evaluation and interpretation of the marker signals (Schulten & Leinweber, 1999).

The DOM's C and N *K*-edge XANES analyses were conducted following the procedures used for C and N *K*-edge XANES analyses from the solid SOM (Purton et al., 2015; Negassa et al., 2019). In brief, we vortexed about 1.5-ml DOM for 30 s for each DOM sample, and then 15 µl of the DOM was poured on gold-coated silicon wafers and allowed to dry for 4 h. This procedure was repeated three times to concentrate the DOM at the surface of the gold-coated silicon wafers. Next, the wafers were affixed to sample plates using conductive C tape (SGE). The C and N *K*-edge spectra were collected at the high-resolution spherical grating monochromator beamline of the Canadian Light Source Synchrotron in the fast scan mode, Saskatoon, SK, Canada. The spectra were collected using partial fluorescence yield with a silicon drift detector (SDD) placed at 90 degrees to the incident beam in the plane of polarization of the incident photon beam (Gillespie et al., 2015). When placed in this position, the elastic scattering contribution to the signal is minimized. For measuring the incident flux (I0), the elastic scattering intensity from a carbon-free (Au coated) blank substrate was collected using silicon drift detector directly above the sample. This position is normal to the plane of polarization of the incident beam and receives the maximum elastic scattering intensity. The energy of C *K*-edge spectra was calibrated using citric acid at 288.7 eV, and the energy of N *K*-edge was calibrated using $(NH_4)_2SO_4$ at 400.8 eV (Gillespie et al., 2008). We collected 50 scans for each soil sample and 100 scans for each blank sample.

The collected spectra were preprocessed by NeXpy v0.12.4 (Python version 3.7) and normalized by the Athena software package (ver. 0.8.56) (Ravel & Newville, 2005). After fitting a background arctangent function using the curve fitting program fityk (version 0.9.8), a series of Lorentzian curves

TABLE 1 The dissolved organic C and total dissolved N concentrations in the surface and subsurface horizons of the rewetted and drained peatland types

Site	Depth	DOC	DIC	TDC	TDN	C:N
	cm	mg L^{-1}				
AD	0–10	77 (14)	2 (0)	79 (14)	28 (0.8)	3 (0.4)
	40–60	126 (42)	3 (1.4)	129 (40)	28 (10)	5 (0.4)
AW	0–20	209 (45)	4 (0.8)	212 (44)	30 (6)	7 (0.2)
	60–80	186 (43)	5 (0.8)	191 (43)	23 (6)	8 (0.2)
CD	0–10	162 (28)	4 (1.3)	166 (29)	25 (4)	7 (0.9)
	40–60	134 (44)	3 (1.4)	137 (45)	17 (8)	9 (0.7)
CW	0–10	105 (19)	4 (1.9)	109 (20)	14 (2)	8 (0.6)
	20–30	119 (13)	2 (0.8)	121 (14)	10 (2)	13 (1.6)
PD	0–10	438 (117)	5 (1.6)	442 (118)	68 (7)	6 (1.0)
	40–60	115 (28)	3 (0.5)	118 (28)	14 (4)	9 (0.4)
PW	0–20	108 (13)	13 (6.3)	122 (19)	19 (3)	6 (0.3)
	60–80	15 (2)	8 (2.5)	23 (4)	2 (0)	11 (2)
P-value		<.0001	.1839	<.0001	<.0001	.829
SE		32	2	32	4	2

Note. Numbers in the parentheses are standard deviation. AD, drained forest peatland; AW, rewetted forest peatland; CD, drained coastal peatland; CW, rewetted coastal peatland; DIC, dissolved inorganic carbon; DOC, dissolved organic carbon; PD, drained percolation mire; PW, rewetted percolation mire; TDC, total dissolved carbon; TDN, total dissolved nitrogen.

were fitted to the spectral features (Gillespie et al., 2011). The main spectral features of C and N *K*-edge-XANES were assigned by comparison with published spectra (Pavlychev et al., 1995; Leinweber et al., 2010; Lehmann et al., 2009; Solomon et al., 2009; Kruse et al., 2011; Negassa et al., 2019). Fitting spectral features with Lorentzian line shapes does not allow the extraction of quantitative concentrations of the different functional groups. Instead, this methodology provides a way to reliably observe the relative differences between the spectra and assign these differences to functional groups based on the energy position.

2.4 | Statistical analyses

The ANOVA of the DOC and TDN concentrations, C/N ratio of the DOM, and DOM compound classes were conducted with the SAS software (SAS 9.4) using PROC GLIMMIX. The different soil depths in different peatland types were treated as the fixed effect, whereas the replication and the soil depth interaction with replication were treated as the random effect. The normality of the data was tested before running any statistical analysis using PROC UNIVARIATE in SAS software, and the Shapiro-Wilk W test for normality was used (Shapiro & Wilk, 1965). Some of the data were log-transformed to fulfill the assumption of ANOVA. However, the least-square means of the original data were presented. We presented the data obtained from the topsoil and bottom soil horizons of each peatland type in the main text because soil parameters were similar in the subsurface horizons of the different peatland types. However, the DOC concentrations and DOM compound classes of the whole soil profiles and the topsoil and bottom soil horizons of the spectral intensities of the C and N functional groups are presented in the Supplemental Materials (Supplemental Tables S2 to S5).

3 | RESULTS AND DISCUSSION

3.1 | Dissolved organic matter

The DOC concentration in the rewetted forest peatland was 2.7 times higher than in its respective drained forest peatland, whereas the DOC concentrations in the rewetted coastal peatland and percolation mire were lower than in their respective drained peatlands by 1.5 and four times, respectively, at the topsoil horizons ($P < .0001$) (Table 1 and Supplemental Table S2). The DOC concentration in the deepest soil horizons of the different peatland types was similar except for the highest DOC concentration in the rewetted forest peatland and the lowest concentration in the rewetted percolation mire (Table 1). In most soil horizons of the different peatland types, the dissolved inorganic carbon (DIC) concentration ranged from 2 to 5 mg L^{-1} except in the rewetted percolation mire (Table 1).

The highest DOC concentration in the rewetted forest peatland compared with the rewetted coastal peatland and percolation mire (Table 1) indicated forest litters' susceptibility

to decomposition during the short summer season high-temperature and conducive soil moisture (Bell et al., 2018). The highest DOC concentration in the subsurface horizons of the rewetted peatland (Supplemental Table S2) also indicated a DOM leaching and translocation from the topsoil horizons to the subsurface horizons in the coarse-textured forest peat (Tuukkanen et al., 2017). Like the soil organic C (C_{org}) concentration in the long-term drained forest peatland (Negassa et al., 2019), drainage substantially decreased the DOC concentration of the forest peatland (Table 1) because of fast DOM turnover and oxidative losses. The DOM's lower C/N ratio (Table 1) than the C/N of the solid peat soil samples (Negassa et al., 2019) further confirmed the faster DOM turnover than the SOM. The mineral soil interlayers in the drained forest and coastal peatlands (Negassa et al., 2019) can also enhance the DOM turnover (Kalbitz, 2001; Dinsmore et al., 2013; Pinsonneault et al., 2016). Furthermore, DOM can leach from the surface horizons and form stable compounds with other elements (McDowell, 2003; Kaiser & Kalbitz, 2012), contributing to the lowest water-extractable DOM concentration from the highly degraded drained forest peatland. A previous study on northeastern Germany's peatlands also reported lower DOC concentration in severely degraded peatlands than slightly degraded ones (Schwalm & Zeitz, 2015).

The lowest DOC concentration in the rewetted percolation mire (Table 1) can be explained by the site's permanent rewetting, resulting in low DOC production that agreed with a similar study from southern Germany (Höll et al., 2009). The DOC concentrations (Table 1) agreed with their C_{org} concentrations (Negassa et al., 2019), particularly at the drained peatlands' surface horizons. As a result, the highest DOC concentration in the drained percolation mire (Table 1) was in line with its highest C_{org} concentration (Negassa et al., 2019). This implies that draining peatlands with the highest C_{org} concentration increases peat decomposition, DOC production, and likely GHG emissions. Except in forest peatlands, the lower DOC concentration in the subsurface horizons of coastal peatland and percolation mires than their respective topsoil horizons indicate minimal peat decompositions in the subsurface horizons because of anoxic soil conditions.

The TDN concentrations in contrasting management practices of the forest peatlands were similar. Still, like the DOC concentration, the TDN concentrations in the drained coastal and percolation mire were higher than in their rewetted counterparts by 1.79 and 3.58 times, respectively, at the topsoil horizons (Table 1). Although the TDN concentration differed among the different peatland types, the TDN was similar between each peatland type's rewetted and drained subsurface horizons. Rewetting increased the C/N ratio of the DOM only in the rewetted forest peatlands, but rewetting did not change the C/N ratio of DOM in the topsoils of the coastal peatlands and rewetted percolation mires.

The lower TDN concentration in the rewetted coastal and percolation mire than in their respective drained peatlands at the surface horizon (Table 1) indicates a lower N mineralization in rewetted peatlands than drained peatlands. External N sources also likely contributed to the higher TDN concentration in the topsoils than in the peatlands' respective deepest soil horizons. For instance, the highest TDN concentration at the drained percolation's surface horizon can be attributed to N fertilization for hay production for many years (Jurasinski et al., 2020). The lower C/N ratio of the DOM (Table 1) than the C/N ratio of the solid SOM (Negassa et al., 2019) indicated the preferential release of TDN and N coming from external sources, confirming findings of previous studies elsewhere (Frank et al., 2014; Kalbitz & Geyer, 2002; Urbanová et al., 2011). Besides retaining N and other elements in plant tissues and peats, rewetted peatlands provide ideal conditions for denitrifiers and possess a great capacity to remove excess nitrate (Schlesinger & Bernhardt, 2013).

The most critical factors contributing to substantial differences in DOC concentrations among the different peatland types were their considerable differences in drainage periods. Although rewetting reduced the DOM concentrations in the coastal and percolation mire, regardless of their respective SOM concentration, the DOM concentration of the rewetted forest was the second to the DOM of the drained percolation mire. No question, rewetting increased SOM and reduced DOM production; however, easily degraded plant material such as forest litter under seasonal soil moisture and temperature fluctuation increased DOM production in rewetted forest peatland of the present study. Groundwater dynamics in different seasons also contributed to peat decomposition and DOM production, typical in all study sites of the present study (Ahmad et al., 2020). Such seasonal groundwater dynamics are specific in peatlands rewetted by passively rising water tables. Overall, the DOM concentrations in the rewetted peatlands depend on peatland types, SOM concentration, drainage and rewetting periods, groundwater dynamics, current vegetation cover, and accumulated litter types. For example, rewetting highly degraded peatlands of northeastern Germany substantially increased DOC concentrations at the inception of rewetting (Zak & Gelbrecht, 2007). However, rewetting different peatland types for about 10 and 20 yr decreased the DOC concentrations in Quebec, Canada, and southern Germany, respectively (Höll et al., 2009; Strack et al., 2015).

The DOC and TDN concentrations determined from the long-term drained and rewetted different peatland types do not account for the total DOM produced during peat decomposition. Because DOM is the source of plant nutrients and energy for soil microbial organisms, losses occur in the form of GHG emissions, sorbed to different mineral matters, percolate to groundwater, and transported to nearby water bodies (Bengtsson & Torneman, 2004; Jennings et al., 2020). Therefore, the lowest DOC concentrations in peatlands with the

predominant inorganic mineral matters of the drained forest peatland and coastal peatlands (Table 1) can be attributed to these DOM influencing factors (Qualls, 2013). However, the lower DOM concentration in rewetted percolation mire can be explained by the lower peat decompositions rate under the anoxic soil conditions. Overall, considering all factors contributing to peat degradation and DOM production, peatland type, and land-use history must be regarded to understand the restoration processes of degraded peatlands and unwanted impacts of peatlands restoration on the terrestrial and aquatic ecosystem.

3.2 | DOM compound classes

The DOM's thermal pyrolysis products and thermal stability between the rewetted and drained forest peatlands were highly different; however, slightly different between the drained and rewetted coastal peatlands and between the drained and rewetted percolation mires (Figure 1). The total ion intensity (TII)-thermograms of the rewetted forest peat samples showed peaks at 250 and 440 ˚C, whereas the peak of TII-thermograms of the drained forest peat soil samples was observed at 340 ˚C. The intense signals and thermograms were similar in the rewetted and drained coastal peatlands and slightly higher in the drained than rewetted percolation mires. The differences in TII-thermograms and pyrolysis products between the rewetted and respective drained peatlands (Figure 1) were in line with their DOC (Table 2 and Supplemental Table S2) and C_{org} concentrations (Negassa et al., 2019). The thermograms of the rewetted forest peatland (Figure 1b) were more than that of the drained forest peatland (Figure 1a), indicating the less stability of DOM of the drained forest peat soils. The DOM thermal stability of the different peatland types was similar to their corresponding SOM thermal stability (Negassa et al., 2019), showing that the DOM molecular compositions are as complex as solid SOM molecular compositions.

The TII, the volatile matter (VM), and the DOM compound classes were different between the rewetted and respective drained peatlands (Table 2 and Supplemental Table S3). The TII differences between the drained and rewetted forest and coastal peatlands were similar to their DOC concentration differences (Table 2). However, the TII was lower in the rewetted (6.65×10^8 counts mg^{-1} soil) than in the drained (8.17×10^8 counts mg^{-1} soil) percolation mires. The VM of the rewetted forest peatland was higher than that of the drained forest peatland, whereas the VM of the rewetted coastal peatland was lower than that of the drained coastal peatland. The VM of the rewetted and drained percolation mires was the same. The VM was considerably higher in the topsoils than in each peatland type's bottom soil horizon.

All the DOM compound classes' relative abundances differed between the rewetted forest peatland and drained forest peatland. The abundances of the carbohydrates (CHYDR), phenols + lignin monomers (PHLM), heterocyclic nitrogen compounds (NCOM), and amides (amino acids, peptides, and amino sugars; PEPTI) were lower in the rewetted forest peatland than in the drained forest peatlands. In contrast, the abundance of lignin dimers (LDIM), suberin (SUBER), and C_{16} to C_{34} free fatty acids (FATTY) were higher in the rewetted than in the drained forest peatlands. The relative abundances of DOM compound classes were similar between the rewetted and drained coastal peatlands. In contrast, only the CHYDR, LDIM, and the PEPTI relative abundances differed between the rewetted and drained percolation mires.

The remarkable differences in relative abundances of the DOM compound classes between the rewetted and drained forest peatlands (Table 2 and Supplemental Table S3) reflected the considerable differences in their DOC (Table 1 and Supplemental Table S3) and C_{org} concentrations (Negassa et al., 2019). The higher abundances of labile DOM compound classes such as CHYDR, PHLM, heterocyclic nitrogen and nitriles, and PEPTI in the drained than the rewetted forest peatlands can be explained by the continuous inputs of easily degradable forest litter, microbial metabolites (especially CHYDR and PEPTI), and degradation of LDIM, SUBER, and FATTY to labile compounds in the former (Leinweber et al., 2001; Kalbitz et al., 2003). The drainage and predominance of soil minerals in the drained forest peatland enhanced oxidative peat degradation and mineralization (Negassa et al., 2019). The lower proportions of these labile organic compound classes in the rewetted forest peatland than in its respective drained peatland can be attributed to the slower peat decomposition under more water-saturated conditions. Thus, the labile DOM compound classes were parts of the partially decomposed peat and were not released to the soil solution under anoxic soil conditions. However, the higher abundances of LDIM, SUBER, and FATTY in the rewetted than in the drained forest peatland indicate that the compound classes perhaps produced before rewetting and have been preserved under the anoxic soil conditions or originate from the residues of plants that have been newly established under the rewetted conditions. Although there is no information on the stability of labile and stable DOM compound classes under anoxic soil conditions, a previous study reported that the sorbed labile DOM compound classes had increased stability compared with the sorbed stable compound classes (Kalbitz et al., 2005).

The lack of differences in relative abundances of the DOM compound classes between the rewetted and drained coastal peatlands and percolation mires also agreed with their respective C_{org} concentration (Negassa et al., 2019). The coastal peatlands and percolation mires were also under

TABLE 2 The relative abundance of the dissolved organic matter compound classes in the drained and rewetted peatlands revealed by Py-FIMS

			Total ion intensity (TII)									%
Site	**Depth**	**TII**	**CHYDR**	**PHLM**	**LDIM**	**LIPID**	**ALKY**	**NCOMP**	**PEPTI**	**SUBER**	**FATTY**	**VM**
	cm	10^6 counts mg^{-1} soil	%									
AD	0–10	44 (2)	12.9 (0.4)	14.4 (0.3)	0.9 (0)	3.4 (0.1)	10.7 (0)	6.8 (0.1)	19.1 (0.7)	0.05 (0)	1.3 (0.1)	67 (2)
	40–60	264 (25)	2.4 (0.1)	13.8 (1.3)	4.5 (0.4)	9.8 (0.3)	22.5 (0.2)	1.0 (0.1)	3.1 (0)	0.01 (0)	4.9 (0.1)	28 (19)
AW	0–20	577 (237)	2.2 (0)	8.2 (0.9)	3.5 (0.1)	13.4 (0.7)	9.6 (0.3)	1.2 (0)	3.3 (0.1)	0.64 (0.1)	4.1 (0)	80 (6).
	60–80	728 (87)	1.9 (0.1)	6.5 (0.1)	4.4 (0)	12.1 (0.2)	9.3(0.1)	0.9 (0)	2.9 (0.1)	0.83 (0)	2.7 (0.2)	53 (2)
CD	0–10	523 (30)	3.0 (0.2)	9.2 (0.3)	4.2 (0.1)	9.0 (0.1)	11.8 (0.3)	1.5 (0)	4.0 (0.2)	0.53 (0.1)	1.7 (0.1)	73 (5)
	40–60	773 (50)	3.1 (0)	8.7 (0)	4.8 (0.1)	9.9 (0.1)	12.6 (0.1)	1.3 (0)	3.6 (0)	0.48 (0)	2.2 (0.2)	52 (6)
CW	0–10	234 (84)	2.9 (0.6)	8.4 (1)	4.5 (0.3)	11.8 (1.1)	11.4 (0.8)	1.6 (0.3)	4.1(0.8)	0.42 (0.1)	1.9 (0.2)	33 (6)
	20–30	135 (16)	3.0 (0.3)	10.1 (0.1)	4.7 (0.3)	10.9 (0.2)	12.1 (0.2)	2.4 (0.2)	3.7 (0.2)	0.33 (0)	1.4 (0.1)	38 (7)
PD	0–10	817 (275)	2.6 (0.4)	8.8 (0.5)	5.2 (0)	9.3 (0.5)	11.2 (0.7)	1.3 (0.1)	3.2 (0.3)	0.69 (0.2)	2.9 (0.3)	76 (5)
	40–60	332 (799	2.3 (0.6)	11.2 (1.5)	6.1 (0.5)	9.8 (0.3)	11.0 (0.9)	1.3 (0.4)	3.3 (0.6)	0.36 (0.1)	2.0 (0.2)	49 (12)
PW	0–20	665 (76)	3.4 (0.2)	10.1 (0.3)	3.8 (0.5)	8.5 (0.1)	11.2 (0.3)	1.5 (0.2)	4.3 (0.2)	0.57 (0)	3.2 (0.2)	78 (9)
	60–80	864 (95)	1.1 (0)	5.1 (0.5)	6.2 (0.3)	8.0 (0.1)	7.0 (0.5)	0.5 (0)	1.9 (0.1)	0.78 (0)	5.6 (0.9)	53 (10)
P-value		0.0012	<.0001	<.0001	<.0001	<.0001	<.0001	<.0001	<.0001	<.0001	<.0001	.651
SE		86	0.2	0.5	0.2	0.3	0.3	0.1	0.3	0.06	0.2	6

Note. Numbers in the parentheses are standard deviation. AD, drained forest peatland; ALKY, alkyl aromatics; AW, rewetted forest peatland; CD, drained coastal peatland; CHYDR, carbohydrates; CW, rewetted coastal peatland; DIC, dissolved inorganic carbon; DOC, dissolved organic carbon; FATTY, free fatty acids C^{16} to C^{34}; LDIM, lignin dimers; LIPID, lipids, alkanes, alkenes, fatty acids, n-alkyl esters; NCOMP, heterocyclic nitrogen and nitriles; PD, drained percolation mire; PEPTI, amides (amino acids, peptides, and amino sugars); PHLM, phenols + lignin monomers; PW, rewetted percolation mire; SUBER, suberin; TDC, total dissolved carbon; TDN, total dissolved nitrogen; VM, volatile matter.

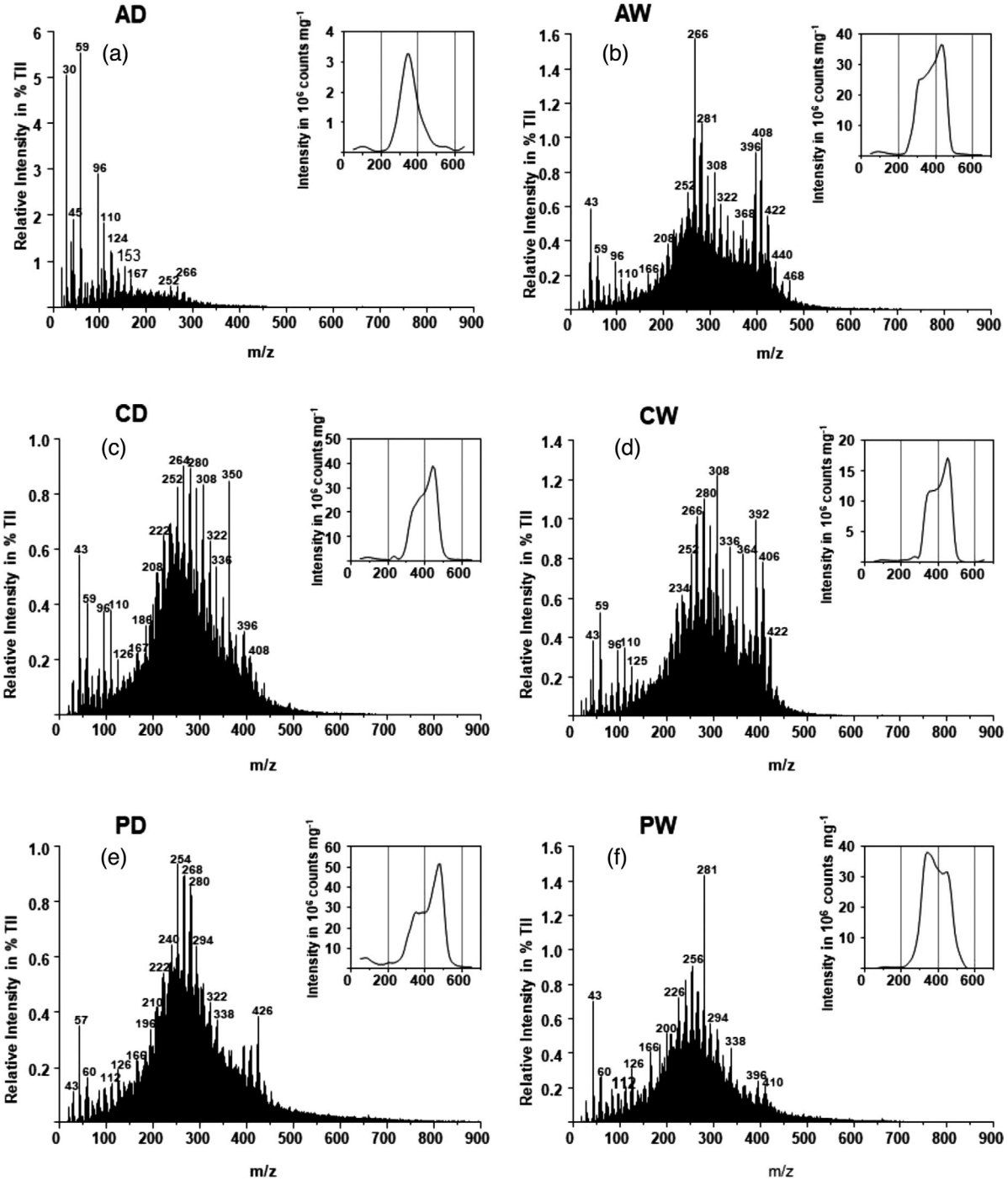

FIGURE 1 Summed and averaged pyrolysis field ionization mass spectra and thermograms of total ion intensity (TII) of the dissolved organic matter extracted from surface horizons of the drained and rewetted peatland types. AD, drained forest peatland (a); AW, rewetted forest peatland (b); CD, drained coastal peatland (c); CW, rewetted coastal peatland (d); PD, drained percolation mire (e); PW, rewetted percolation mire (f)

a similar vegetation cover and land uses that could contribute to stable and bio-resistant organic compound classes mainly originating from refractory plant residues such as lignin, tannins, and polyphenols (Dodla et al., 2012; Lin & Simpson, 2016; Lv et al., 2020). The present study results indicated that the relative abundances of the DOM compound classes depended more on the intensity of peat degradation than the DOC concentrations. Hence, permanent rewetting enhances the accumulation of newly formed and old DOM in peatland ecosystems.

3.3 | DOM organic matter functional groups

The C *K*-edge XANES analysis revealed nine C functional groups with various spectral intensities (Figure 2 and Supplemental Table S4). Spectral features of the different C

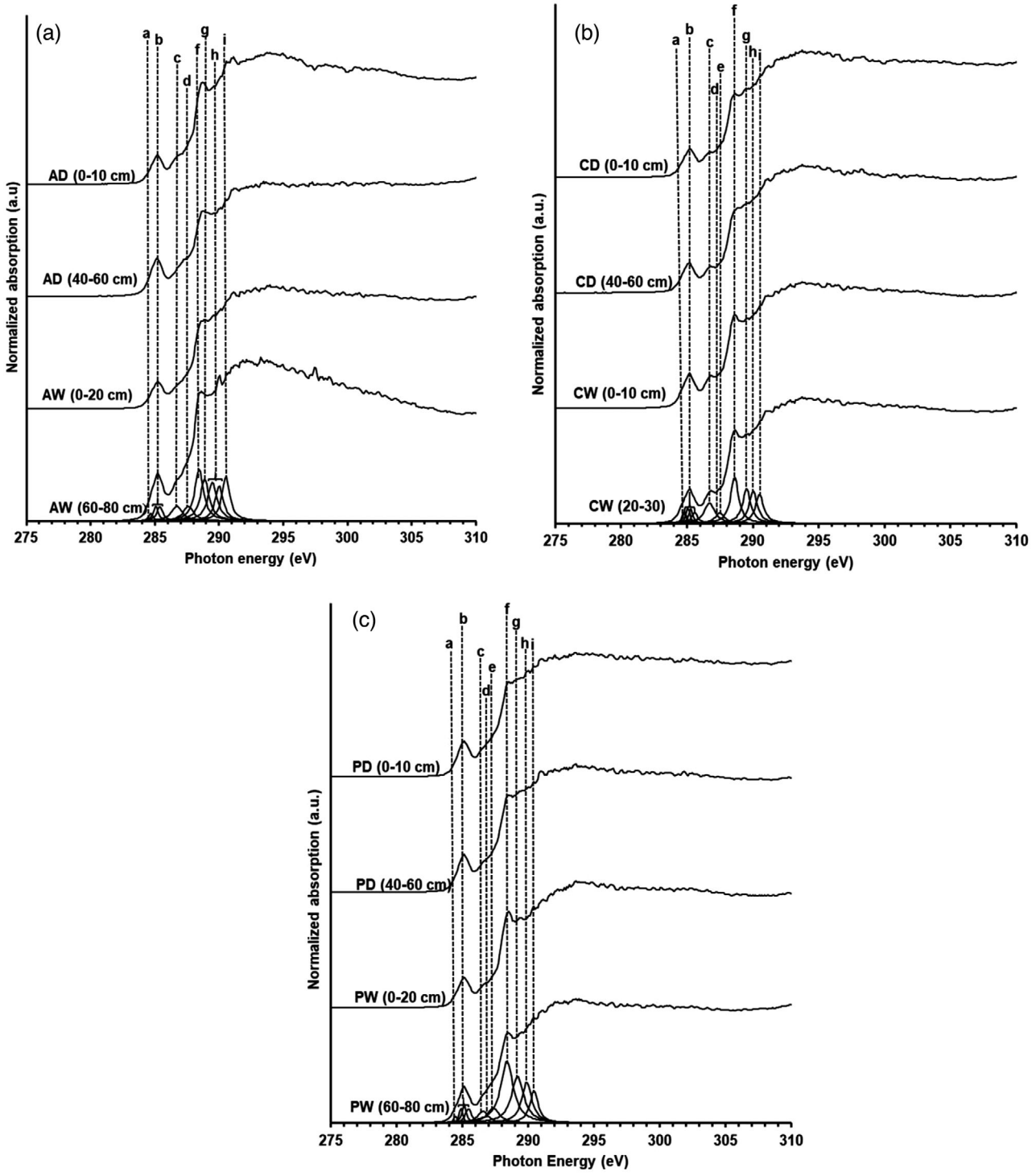

FIGURE 2 Normalized C *K*-edge X-ray absorption near-edge structure (XANES) spectra of dissolved organic matter (DOM) extracted from different drained and rewetted peatland types. (a) forest peatlands, (b) coastal peatlands, (c) percolation mires. AD, drained forest peatland; AW, rewetted forest peatland; PD, drained percolation mire; PW, rewetted percolation mire; CD, drained coastal peatland; CW, rewetted coastal peatland. Numbers in parentheses next to the abbreviation of the names of the study sites indicate soil sampling depths in cm. Spectral features were indicated by a: *p*-benzoquinone (284.5–284.8 eV); b: double C bound in protonated and alkylated aromatic and alkenes (284.9–285.5 eV); c: carbonyl C bound to ketone (286.6–286.8 eV); d: aromatic C bound to hydroxyl and ether (287.1–287.3 eV); e: aliphatic C bound to amino acids and phospholipid fatty acids (287.4–287.6 eV); f: carboxylic C bound to amide group and acetate (288.2–288.6 eV); g: aliphatic C bound to –COOH (288.9–289.2 eV); h: O-alkyl C bound to polysaccharides, protein, nucleic acid, alcohols and ethers (289.5–290.0 eV); i: carbonate (290.1–290.5 eV)

functional groups in the rewetted and drained forest peatlands of the surface horizons were similar; however, the spectral intensities of most C functional groups differed between the topsoil and bottom soil of the drained forest peatland (Figure 2a, Supplemental Table S5). Accordingly, the spectral intensities of the carbonyl C bound to ketone (286.6–286.8 eV, feature c), aliphatic C bound to amino acids, and phospholipid fatty acids (287.4–287.6 eV, feature e), and O-alkyl C (289.5–290 eV, feature h) were stronger in the rewetted coastal peatland than in the drained coastal peatland at the topsoil horizons. Most C functional groups' spectral intensities were similar in the rewetted and drained coastal peatlands' topsoils and bottom soil horizons. The C functional groups' spectral intensities such as features d, g, h, and i were different in the contrasting management practices of the percolation mires, whereas the remaining C functional groups' spectral intensities were similar. The C functional groups' spectral intensities designated as features e, f, g, h, and i were the predominant groups regardless of differences in management practices, soil horizons, and peatland types.

The differences in C functional groups between the rewetted and respective drained peatlands were minimal compared with their DOC (Table 1 and Supplemental Table S2) and C_{org} concentrations (Negassa et al., 2019). Unlike the Py-FIMS (Table 2), the C *K*-XANES was not sensitive enough to distinguish the degree of peat degradation and influence of rewetting and drainage peatlands on C functional groups. The Py-FIMS detects the molecular weight of thermally volatilized organic compounds, whereas the XANES probes the electronic environment of an atom in a molecule. The highest spectral intensities of the carboxylic C bound to amide group and acetate, O-alkyl C, and aliphatic C bound to −COOH (Figure 2 and Supplemental Table S4) of the DOM similar to the spectral intensities of the C functional groups of the peat soils (Negassa et al., 2019). The predominance of these functional groups were also reported for the DOM of peat soils revealed by ^{13}C NMR (Moody et al., 2018). The carbonyl C bound to a ketone (Figure 2, feature c) indicated microbial metabolized organic compounds in the DOM (Gillespie et al., 2014). The lack of aromatic C bound to hydroxyl and ether in most peat soils (Figure 2, feature d and Supplemental Table S4) can be attributed to the partial peat decomposition did not result in DOM with such stable high molecular weight aromatic C functional group (Hansen et al., 2016).

The Py-FIMS and XANES analyses showed that DOM is as complex as SOM in soils of different peatland types. Furthermore, the similarity in C functional groups in the DOM of the different peatland types can be attributed to the extracted DOM was similar in degradation status regardless of the materials' variations and ages (Zark & Dittmar, 2018). In addition, the role of the C-functional group in cation exchange capacity and the complexation of trace metals could prevent DOM degradation by microbial organisms, thereby hinder the contribution of DOM to GHG emissions in terrestrial and freshwater ecosystems (Ellerbrock & Kaiser, 2005; Lau & del Giorgio, 2020).

The N *K*-XANES analysis revealed seven N functional groups in the DOM extracted from the drained and rewetted peatland types; however, some N functional groups were absent in some soil horizons and peatland types (Figure 3 and Supplemental Table S5). Differences in signal intensities of most N functional groups between the rewetted and drained forest peatlands were weaker than the differences between the topsoil and bottom soil samples of each management practice of each peatland type. The spectral intensities of N functional groups (features a, c, and d) were weaker in the rewetted than in the drained coastal peatlands; however, the spectral intensities of features b, e, and f of the surface horizons of the drained and rewetted coastal peatlands were similar (Figure 3b). The spectral features of the bottom soil horizon of the rewetted coastal peatland were stronger than its topsoil horizons, whereas the spectral intensities of most N functional groups in the topsoil horizons of the drained coastal peatlands were stronger than its respective bottom soil horizons. Similarly, spectral features of the N functional groups were stronger in the rewetted than the drained percolation mire at the topsoil horizon except for the features b and f (Figure 3c). The spectral feature of nitrate-N was observed only in the drained forest and rewetted coastal peats and percolation mires, whereas ammonium was not detected in any peat horizon.

Lack of consistencies in N functional groups' spectral intensities between topsoil and bottom soil horizons and between drained and rewetted peatlands can be attributed to the variation in TDN concentrations between the drained and rewetted peat soils, particularly at the topsoil horizons (Table 1). A previous study reported the predominant N molecular compositions in the soil solution were free amino acids (Warren, 2014); however, the results of XANES analysis in the present study revealed that the DOM contained heterocyclic N functional groups in addition to amino acid fragments and amine group (Figure 3). The nitro compounds were the predominant N functional group in all peatland types in the present study (feature f, Figure 3, and Supplemental Table S5), whereas the amide group was the predominant N functional group in the solid peat soils of different peatland types (Negassa et al., 2019). The substantial differences in the spectral intensities between each peatland type's topsoils and subsoil horizons compared with the different peatland types can be attributed to N turnover in the topsoils than subsoil horizons, particularly during the summer season when groundwater tables are lower than in other seasons (Kalbitz et al., 2002). The similarity in N functional groups among the DOM extracted from different peatland types can be explained by the sensitivity of most N compounds to biodegradation, resulting in similar N functional groups (Li et al., 2018). The N *K*-XANES analysis revealed more N functional groups and

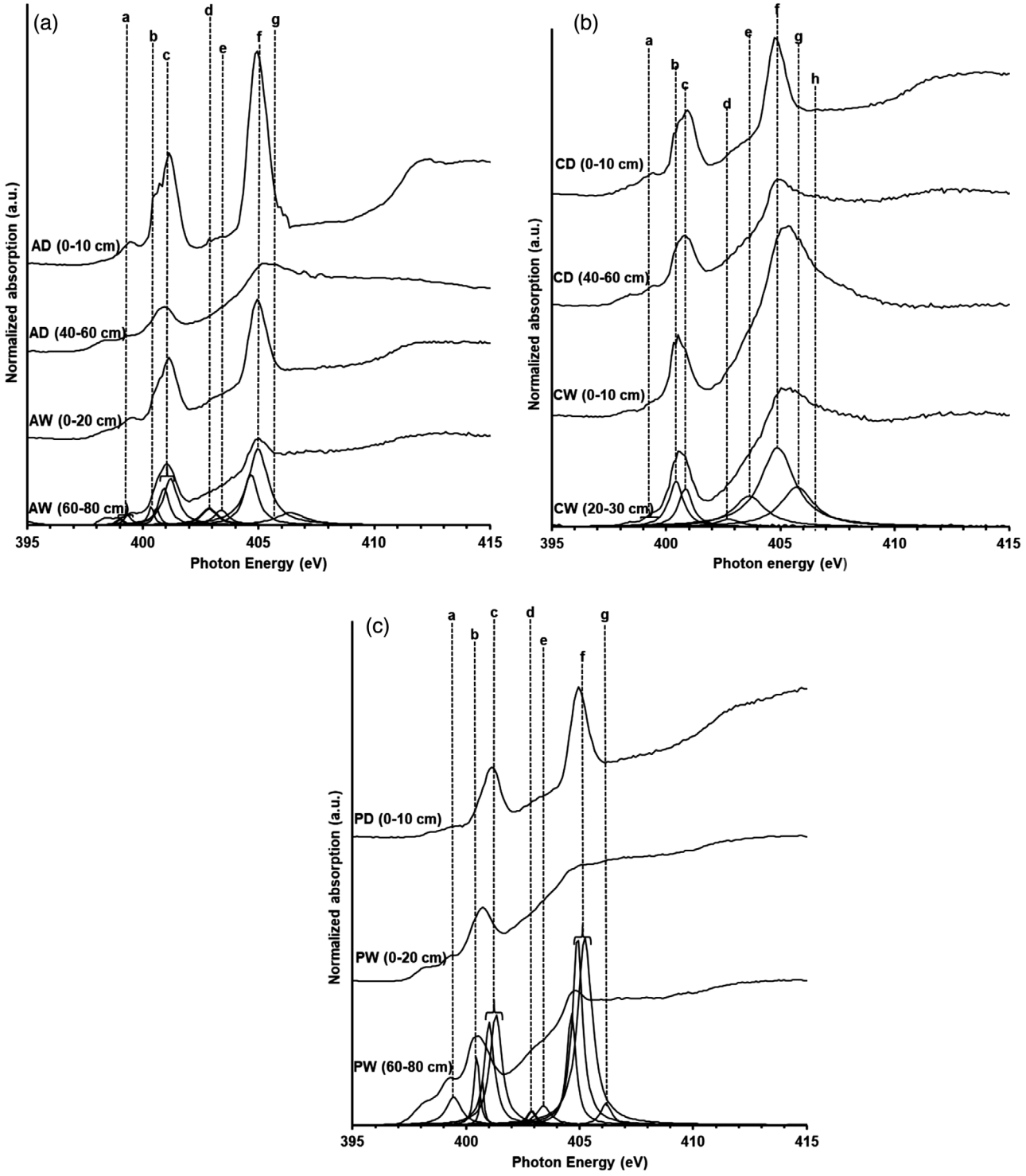

FIGURE 3 Normalized N *K*-edge X-ray absorption near-edge structure (XANES) spectra of dissolved organic matter DOM extracted from different drained and rewetted peatland types. (a) forest peatlands, (b) coastal peatlands, (c) percolation mires. AD, drained forest peatland; AW, rewetted forest peatland; PD, drained percolation mire; PW, rewetted percolation mire; CD, drained coastal peatland; CW, rewetted coastal peatland. Numbers in parentheses next to the abbreviation of the names of the study sites indicate soil sampling depths in cm. Spectral features were indicated by a: amino acid decomposition products (399.0–399.7 eV); b: pyridone and cytosine (400.5–400.9 eV); c: amide (401.0–401.3 eV); d: pyrrole and uracil (402.6–402.9 eV); e: 4N bonded aromatic (403.3–403.8 eV); f: nitro compounds (404.4–404.8 eV); g: inorganic NO_3^- (405.7–405.9 eV)

better spectral quality in the DOM than in the solid SOM (Jokic et al., 2004; Gillespie et al., 2014). This is because the solution and liquid samples dried at the silicon wafers yielded more homogeneous samples than suspended powders of solid materials usually used for XANES analysis in bulk soil samples (Calvin, 2013).

The absence of nitrate-N features in six of the 12 soil horizons (Figure 3 and Supplemental Table S5) could be attributed to its high mobility in the drained and mineral matter-dominated peatlands and low peat decomposition in the rewetted percolation mire. This result disagreed with a DOM of a mineral soil where nitrate-N was the predominant N functional group (Kruse et al., 2010). This tremendous difference between the mineral and organic soils in nitrate-N feature is not surprising because the former was influenced by heavy N fertilization with <1% C_{org} concentration. The complete absence of an ammonium-N feature in the DOM of all sites indicates cold water cannot extract ammonium, which is usually extracted by basic chemical solutions (Maynard et al., 2008). Although the spectral intensities of different C and N functional groups differed in different soil horizons and peatland types, the influence of drainage and rewetting on the functional groups was not apparent.

3.4 | Synthesis

The negative impacts of drainage and the benefits from restoring degraded peatlands on ecological, economic, and cultural issues have been known (Joosten et al., 2017). As a result, the influences of drainage and rewetting on the biogeochemistry of fen and bog peatlands have got attention for the last two decades (e.g., Andersen et al., 2013; Abdalla et al., 2016; Kitson & Bell, 2020). Although most previous studies have focused on the biogeochemistry of fen and bog peatlands, each peatland type could comprise peatlands different in pedogenesis, drainage period, previous and current vegetation cover, and land use types that influence the response of biogeochemical processes to long-term drainage and rewetting practices. Therefore, it would be difficult to understand the impact of drainage and rewetting on the biogeochemistry of different peatlands without considering these vital peat characteristics in a given peatland ecosystem. As a result, some studies reported undesired effects of peatland restoration by rewetting, such as increasing DOM concentration (Herzsprung et al., 2017), P mobilization (Zak et al., 2010), and greenhouse gas emissions (Rigney et al., 2018). Interpretation of such results requires precaution since short-term rewetting can enhance biochemical processes that increase DOM concentration, P mobilization, and greenhouse emissions because of carry-over processes from the previous long-term drainage that enhanced peat degradation.

In the present study, the DOC concentrations in the drained peatlands (Table 1) were directly proportional to their C_{org} concentrations (Negassa et al., 2019). However, the DOC concentration was the lowest in the permanently rewetted percolation mire, whereas the DOC concentration was the highest in the rewetted forest peatland compared with the other rewetted peatlands (Table 1) because of continued inputs of easily degradable plant materials that mineralized during the summer season. Except at the rewetted percolation mire, the groundwater tables lowered below 50 cm during the summer at the rewetted forest peatland and coastal peatlands (Ahmad et al., 2020). Such lowered groundwater table could facilitate soil microbial activities to mineralize forest litter during the summer (Broder et al., 2017). Although rewetting may increase methane emissions, the carbon sequestered by rewetting can substantially offset the methane emission in a long-term peatland rewetting (Günther et al., 2020).

Furthermore, classifying peatlands to only fens and bogs could ignore other inherent peatland characteristics and anthropogenic effects on peatland biogeochemistry when subjected to long-term drainage and rewetting, as we can conclude from the different properties of fen peatlands of the present study (Negassa et al., 2019; Jurasinski et al., 2020). Indeed, the long-term drainage and rewetting impact on the DOC concentrations and molecular chemical composition in three fen peatland types reflected these two effects. Among the three pairs of hydrologically different peatlands—all belonging to fen peatlands—the drainage and rewetting substantially influenced the DOC concentrations and molecular-chemical composition, particularly in the forest peatlands. We explain this by drainage of the forest peatlands for more than 200 yr, causing subsidence of about 1 m (Weil et al. 2020) and a substantial depletion of the C_{org} concentration (Negassa et al., 2019).

The substantial differences in the DOM molecular-chemical compositions between the management practices and peatland types were mainly related to peatland degradation status. As a result, the DOM molecular-chemical compositions of the highly degraded, drained forest peatland were quite different from the DOM molecular-chemical compositions of the other peatlands. Regardless of slight differences in labile DOM compound classes (CHYDR, PHLM, NCOM, and PEPTI) between the drained and rewetted pairs of the coastal peatlands and percolation mires, the proportions of stable DOM compound classes were similar in all peatland types. Furthermore, the influence of long-term drainage and rewetting peatlands was not reflected by the C and N functional groups of the DOM identified by XANES (Figures 2 and 3). The lack of substantial differences between the drained and rewetted peatlands in molecular chemical composition and functional groups among most peatlands (Table 2, Figures 2 and 3) indicated the similarity in DOM molecular compositions and functional groups

among the different peatland types. Similar results of lacking differences in DOM quality were also reported for differently managed peatlands (Herzsprung et al., 2017; Peacock et al., 2018). Overall, the permanent rewetting reduced the DOC concentrations compared with the drained peatlands in most peatlands; however, the DOC concentrations in all peatland types can contribute to GHG emissions and freshwater pollutions. Thus, understanding the concentration of DOC spatial and temporal dynamics under controlled hydrology, specific for each peatland type, is paramount for monitoring the environmental implications of peatland restoration.

4 | CONCLUSIONS

The methodological approach to combining elemental analyses, C and N XANES, and Py-FIMS was suitable for disclosing molecular composition and functional group differences in DOM of different fen peatland types. The Py-FIMS recorded greater differences than XANES, which is plausible because XANES probes the atoms' chemical environment that may be repeated many times in a molecule, regardless of identical spectral patterns. The same repetitions in the chemical environment of probed atoms like C and N essentially result in enormously different molecular weights recorded by Py-FIMS. Therefore, the higher discrimination power of Py-FIMS than C- and N-XANES is highly plausible. The absence of a clear and consistent trend for the impact of rewetting on the DOM elemental concentration and chemical composition confirms the outcomes of the same analyses at the solid samples and P speciations. We conclude that other site-specific factors strongly modulate the rewetting effects on conserved or newly formed peat profiles. These factors can be pedogenesis, vegetation, profile depth, degradation status, and rewetting intensity. Therefore, predictions of rewetting success and possible effects on adjacent ecosystems must consider all these confounding factors, and generalizations should be made with great caution in each peatland type for scientific monitoring of risk assessment.

DATA AVAILABILITY STATEMENT

All data presented in this manuscript can be accessed at https://www.wetscapes.uni-rostock.de/en/data-portal.

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AUTHOR CONTRIBUTIONS

Wakene Negassa: Conceptualization; Data curation; Formal analysis; Investigation; Methodology; Writing-original draft. Kai-Uwe Eckhardt: Data curation; Methodology. Tom Regier: Data curation; Methodology. Peter Leinweber: Conceptualization; Funding acquisition; Investigation; Project administration; Resources; Supervision; Writing-review & editing.

CONFLICT OF INTEREST

The authors declare no conflict of interest.

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