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Nitrogen Cycling Under Conifer-to-Broadleaf Forest Conversion in Eastern England

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ABSTRACT

In the early 20th century, political focus on timber production in Europe led to extensive afforestation and replacement of broadleaves with often non-native coniferous species. Recent interest in the reverse has risen due to recognition of the wider ecosystem services delivered by forests, such as water quality improvement. Concurrently, recognition of nitrogen saturation in forest soils driven by historically elevated atmospheric deposition has stimulated interest in management interventions aimed at mitigating deposition effects. This study investigated a pseudochronosequence of stands undergoing such conifer-to-broadleaf conversion to capture its effects on soil N cycling in Thetford Forest, UK. The pseudochronosequence covered (1) mature broadleaf stands ($n=5$), (2) mature *Pinus sylvestris* stands ($n=5$), (2) sites felled during the monitoring period ($n=3$) and clearfell sites planted with broadleaves: (3) 0–2 years ago ($n=3$), (4) 5–8 years ago ($n=5$) and (5) 10–13 years ago ($n=5$). The soil C:N ratio at 0–10 cm depth was significantly higher in mature pine sites than in all broadleaf stages. The mean total deep soil NO_3-N leaching fluxes during the study period were lower in young (14.53–16.11 kg-N ha^{-1}) and mature broadleaved stands (10.72 kg-N ha^{-1}) than in mature conifers (23.81 kg-N ha^{-1} $year^{-1}$). However, soil NO_3-N leaching fluxes were not significantly different between forest management stages. Net nitrification rates at 10–30 cm depth were higher under low pH conditions, but soil pH and nitrification rates were not linked to soil NO_3-N leaching fluxes or forest management stages. Although no significant effects of management on NO_3-N leaching were found, this study suggests the need to explore the interactive effects of site characteristics, deposition and forest management impacts on soil processes. Long-term studies complementing observations such as those presented here are needed to capture the effects of conifer-to-broadleaf conversion on forest soil nitrogen dynamics.

1 | Introduction

Elevated nitrogen (N) deposition in the latter part of the 20th century has led to nitrogen saturation in forest soils across Europe (Dise et al. 2009; Macdonald et al. 2002) and other parts of the Northern Hemisphere (Zhu et al. 2015). Nitrate (NO_3-N) leaching occurs where N inputs exceed the total biological ecosystem demand (Aber et al. 1989), contributing to eutrophication and acidification of surface and groundwater (Gundersen et al. 2006; Wright et al. 2001), loss of soil base cations (Waldner et al. 2015) and mobilisation of aluminium, which causes toxicity to roots (Vanguelova et al. 2005) and nutrient imbalances and deficiencies in trees (Jonard et al. 2015).

Due to their all-year canopy and thus a greater atmosphere scavenging ability, coniferous forests are typically associated with higher levels of throughfall ammonium (NH_4-N) than broadleaved forests (Vanguelova and Pitman 2019). Higher NO_3-N leaching fluxes in conifer systems than broadleaves have been attributed to higher throughfall NH_4-N fluxes in

Summary

- Interest in conifer-to-broadleaf has grown across Europe in recent decades.
- Conifer-to-broadleaf conversion is expected to reduce NO_3-N leaching to water bodies.
- Soil C:N ratios at 0–10 cm depth were significantly affected by conifer-to-broadleaf conversion.
- Effects of conifer-to-broadleaf conversion NO_3-N leaching were not observed.

conifers (Rothe et al. 2002). Despite continuing reductions in air pollution across Europe (Dirnböck et al. 2018; Fowler et al. 2007), localised sources from, for example, livestock sheds still contribute to high NH_3 deposition (Vanguelova and Pitman 2019) and increase soil solution N concentrations in forest ecosystems (Stevens et al. 2016). Inputs of N to forest soils accumulate in organic horizons in the first decades, when N is either immobilised or cycled between soil and vegetation, while deeper mineral soils become a more dominant sink in later years (Veerman et al. 2020). Other deposition inputs, such as dissolved organic carbon (DOC), which can originate from natural (e.g., pollen) and anthropogenic sources, also stimulate soil N transformation processes, with throughfall DOC fluxes previously linked to soil NO_3-N leaching fluxes (Pitman et al. 2010; Solinger et al. 2001).

UK observations indicate an accumulated stock of soil N under conifers which, if disturbed by forestry operations, could leach out (Vanguelova and Benham 2024). Even in forests unaffected by high N deposition, harvesting and site preparation operations initially increase NO_3-N leaching fluxes from forest soils. There are several mechanisms behind this, such as soil disturbance, changes in vegetation uptake and altered rates of soil biochemical processes, particularly increased rates of nitrification (e.g., Olsson et al. 2022; Huber et al. 2010). Reduced interception and decreased transpiration due to stand removal result in increased soil drainage and groundwater recharge (Kubin et al. 2017) and increasing groundwater NO_3-N concentrations (Mannerkoski et al. 2005; Neal et al. 2004). However, increased water fluxes can also dilute water pollutants, which has been observed in some cases (Schleppi et al. 2017; Calder et al. 2002). Whilst NO_3-N leaching has been widely studied in the context of afforestation, the effects of changing the forest type on this process remain unstudied, even though converting storm-endangered Norway spruce to oak woodlands has become common in Belgium and Germany from the late 90s onwards (Kint et al. 2006; Kenk and Guehne 2001).

The UK has 1,566,000 ha of coniferous forest, accounting for 48% of forested land (Forest Research 2023a, 2023b), dominated by non-native plantations. The UK Government acquired low-grade agricultural land to plant softwoods at the end of World War I (Raum 2020), and the policy continued until the 1980s (Forestry Commission 1985). To reverse the trend, the Government introduced intentions to restore coniferous plantations on ancient woodland sites to original tree composition in 2005, strengthening those policies in 2022 (Department for Food, Environment and Rural Affairs 2022). To illustrate the challenge, about 223,000 ha of UK conifer woodland was recorded as planted on ancient woodland sites in 2009, qualifying for economic incentives for restoration (Latham et al. 2018). Scenario-led modelling in Wales predicted that 7480 ha of coniferous forest would be converted to broadleaved woodland between 2021 and 2030 (Manzoor et al. 2019).

In the wider forest around the study sites here, increasing the proportion of broadleaves to enhance soil health and support biodiversity is a part of the strategy to increase forest resilience. Restocking clearfelled conifer plantations with broadleaved species changes several components of the forest ecosystem biochemical cycle, including: (1) litter composition, a factor which drives variation in organic and mineral topsoil carbon and nitrogen stocks, soil pH and microbial community composition (Cruz-Paredes et al. 2023; Feng et al. 2022; Ribbons et al. 2018; Cools et al. 2014), (2) soil microbial community activity due to changes in soil temperature and moisture (Brockett et al. 2012; Schindlbacher et al. 2011) and root exudation, (3) plant nutrient demands and mycorrhizal associations (Gao et al. 2020; Liese et al. 2018) and finally, (4) throughfall chemistry (Verstraeten et al. 2023).

The present study aimed to investigate the impact of conifer-to-broadleaf conversion on soil nitrogen cycling processes. The objectives were twofold, to quantify the effects of conifer-to-broadleaf conversion on NO_3-N leaching fluxes in a temperate forest ecosystem using a pseudochronosequence study design, covering the first decade post-conversion and to explore the relationships between NO_3-N leaching fluxes and variables such as throughfall chemistry, soil pH, net nitrification and soil carbon and nitrogen stocks and ratios across the pseudochronosequence. This approach aimed to enhance understanding of the mechanisms driving NO_3-N leaching fluxes during conifer-to-broadleaf conversion.

We hypothesised that (H1) deep soil NO_3-N leaching fluxes will remain elevated in the first decade after planting broadleaves on former coniferous stands, (H2) topsoil nitrification rates are increased following conifer-to-broadleaf conversion, driving an increase in NO_3-N leaching and a decrease in topsoil pH, (H3) differences in throughfall chemistry (DOC, inorganic nitrogen) between forest management stages will explain variation in soil nitrification rates, soil pH and NO_3-N leaching.

2 | Materials and Methods

2.1 | Site Description

The sites used for this study are located in Thetford Forest in the East of England, UK (Figure 1). Thetford Forest is a commercial plantation dominated by *P. sylvestris* (L.) or *P. nigra* subsp. *laricio* but with interspersed broadleaved stands, particularly along woodland block edges. Atmospheric N deposition in the forest is high due to intensive farming in the surrounding area (Environment Agency 2019). A *P. sylvestris* stand at Thetford has been monitored as part of the International Co-operative Program on the Assessment of Air Pollution Effects on Forests (ICP Forests) Level II network since the 1990s (Vanguelova et al. 2007) and shows elevated concentration of NO_3-N in

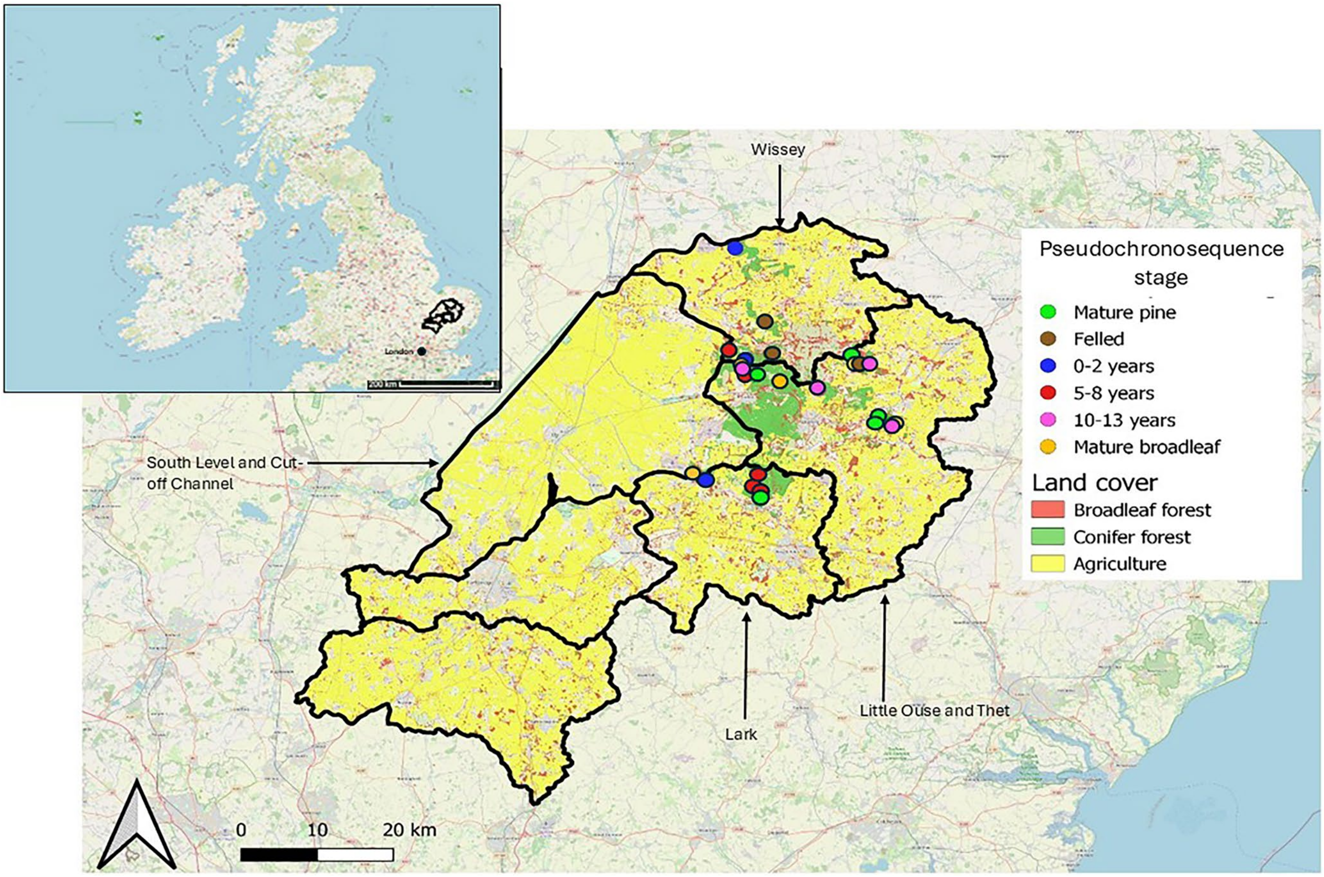

FIGURE 1 | The location of the pseudochronosequence sites across Thetford Forest, East England. Land cover was obtained from UKCEH 2015 landcover map (Rowland et al. 2017), the most distant sites are 37 km apart. The pseudochronosequence stages include: (1) mature pine, (2) recently felled sites, (3) former pine sites planted with broadleaf species: 0–2 years ago, (4) 5–8 years ago, (5) 10–13 years ago and (6) mature broadleaf sites. The management catchment shown is the Cam and Ely Ouse, the sites are located in the operational catchments of Wissey, Lark, Little Ouse and Thet, and South Level and Cut-Off Channel.

deep soil solution, above the 11 mg L^{-1} accepted drinking water standard (Vanguelova et al. 2010). Thetford experiences an average annual rainfall of 600 mm, an annual mean temperature of 11.3°C and an average nitrogen deposition of 14.7 kg ha^{-1} a^{-1} (Vanguelova and Pitman 2019). Thetford Forest frequently experiences droughts, which cause spikes in NO_3-N leaching upon rewetting (Vanguelova et al. 2010). The selected sites are in the Cam and Ely Ouse Management catchment, distributed across the Operational Catchments of the Little Ouse and Thet, Lark, Wissey and South Level and Cut-off Channel (Figure 1). The site overlies a chalk aquifer, which is heavily abstracted for public and agricultural water supply (Supporting Information S1); protecting groundwater quality is thus of stakeholder interest.

A pseudochronosequence of sites was selected to observe the changes in NO_3-N leaching in the first 10 years after converting conifers to broadleaves, consisting of (1) five mature *P. sylvestris* stands, including the Thetford ICP Forests long-term monitoring site, (2) three sites felled in September 2021, February 2022 and March 2022, using conventional harvest with 10% retention, (3) three clearfell sites planted with broadleaves 0–2 years ago, (4) five clearfell sites planted with broadleaves 5–8 years ago and (5) five clearfell sites planted with broadleaves 10–13 years ago, (6) five mature broadleaf sites, planted before 1950 to allow comparison with data in Vanguelova and Pitman (2019). All sites in the younger broadleaf categories were formerly planted with *P. sylvestris* or *P. nigra*. Although the practice is growing, the conversion of conifers to broadleaves is a relatively recent practice. Consequently, identifying a study area within the UK with suitable replicates within age categories was a challenge, so flexibility with broadleaved species was adopted. Sites replanted with naturalised or native broadleaf species were selected preferentially, but non-native broadleaved species (*Alnus cordata* (*LOISEL.*) *DUBy.*, *Quercus rubra* (L.)) were included where sufficient native replicates could not be located. Monocultures were preferred to avoid mixed-species effects. The management history of the mature broadleaves is not entirely clear as detailed records of management history from the 1930–50s are not readily available. Due to the uncertainty, the mature broadleaf stands are used as a reference for the target state of conifer-to-broadleaf conversion, rather than a true representation of broadleaf systems 80+ years after conversion from conifers. Due to the variation in species, the study design is referred to as a pseudochronosequence rather than a true chronosequence. The soil texture across all sites was sandy, and the soil types across the study area are arenosols (IUSS Working Group WRB 2022). Soil horizon formation was poor across sites, with thin organic horizons < 10 cm thick, an A horizon 10–20 cm deep, a weakly developed B horizon and chalk bedrock typically encountered at around 1 m deep. Where the organic layer was present, the humus form was generally moder. Further details of all sites are provided in Supporting Information S2.

2.2 | Sample Collection and Analysis

An overview of the sampling regime is given in Figure 2. NO_3-N leaching flux is a product of soil water flux and NO_3-N concentration. The chloride (Cl) mass balance method is commonly applied to calculate soil drainage fluxes (Svensson et al. 2012). There have been no historical anthropogenic inputs to Thetford Forest, meeting the method's assumption of atmospheric input of chlorine only. Inaccuracies in the method occur in mountainous terrain due to variations in infiltration and runoff (Guan et al. 2010), where a steady state is required (Crosbie et al. 2018). East Anglia is a flat region, minimising this concern. Changes in vegetation cover and site preparation affect pre- and post-felling steady states and can contribute up to 24% of ion variance (Guan et al. 2013). Due to the changes in site typology during felling operations, ground preparation and planting in the initial phases of the conversion process, water fluxes were not calculated for the felled and 0–2 year sites, due to the likely violation of the steady-state assumption in these stages. The chloride mass balance method has previously been compared against a water balance modelling approach at the ICP Forests Level II site at Thetford Forest, confirming the suitability of the approach.

2.2.1 | Throughfall Volume and Chemistry

The purpose of measuring throughfall chemistry was (1) to gather data on Cl inputs for the calculation of water fluxes via the chloride mass balance method, (2) to investigate how variation in throughfall chemistry between pseudochronosequence stages drives deep soil NO_3-N leaching. We therefore measured throughfall chemistry at sites where NO_3-N leaching could be calculated, that is, at sites older than 2 years. At each site, five throughfall collectors were constructed in July 2021. Sampling points were situated along a transect directed towards the centre of the stand, spaced 20 m apart. We left at least a 20 m buffer zone between the edge of the forest and the first sampling point at each site. White PVC funnels ($d = 25$ cm) were mounted using a wooden stake 1 m above ground level and connected via a PVC pipe to a 5 L plastic jerry can buried in the soil or covered with ground vegetation to protect it from direct sunlight. At the ICP Forests Level II site, throughfall chemistry was monitored using the existing collectors, and following the ICP Forests Manual for Sampling and Analysis of Deposition (Clarke et al. 2022). Across the monitoring period, there was no significant difference between the throughfall volume collected at the Level II site compared to the other mature pine sites. Throughfall samples in all 20 sites were collected monthly until March 2022. To manage resource limitations the number of sites was then reduced, and representative sites for each pseudochronosequence stage were selected to continue monitoring until July 2022 ($n = 13$). The collection did not happen in November and December 2021 due to COVID-19 restrictions.

Throughfall samples were transported in a cooler box with ice packs and kept in a refrigerator at 4°C until analysis, as per the ICP Forests protocols (Clarke et al. 2022). Throughfall samples were aggregated for each site and month and filtered through a 0.45 μm mixed cellulose ester filter (Advantec). All throughfall samples August 2021–July 2022 were analysed for Cl and NO_3-N with a Dionex DX 500 Ion Chromatography System with AS 40 autosampler (Sunnydale, USA). Procedural blanks processed using ultrapure water showed nitrate concentrations below or close to the limit of detection. The samples collected in January

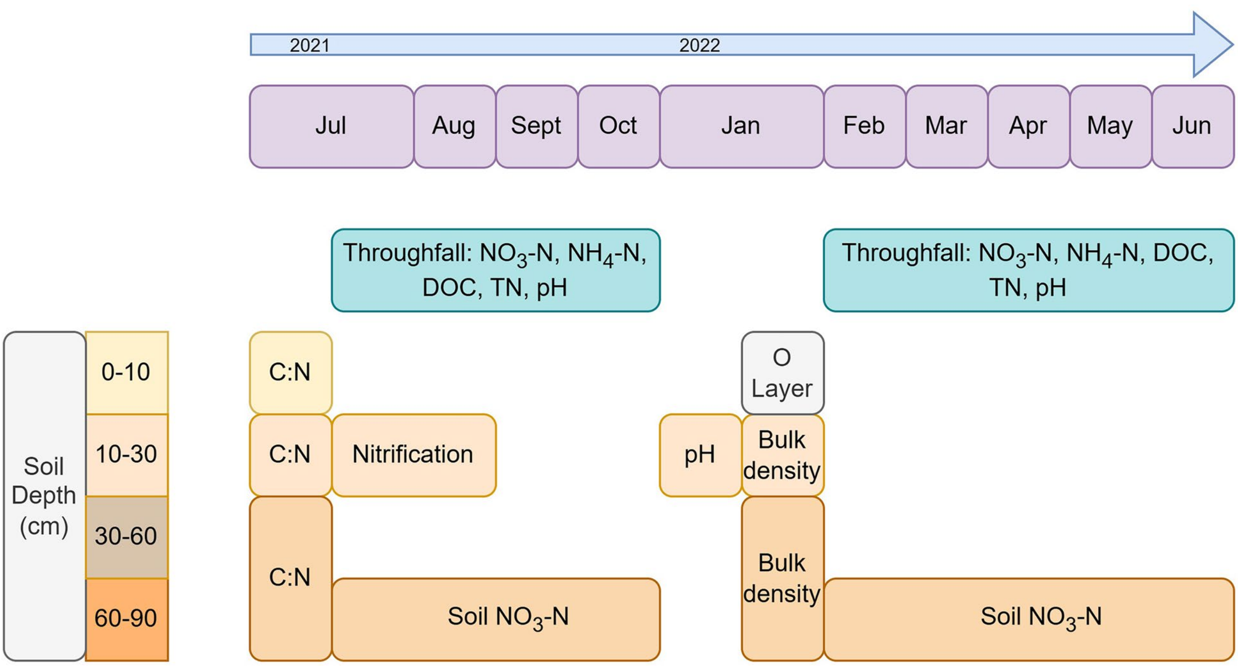

FIGURE 2 | The sampling regime conducted at the study sites in Thetford Forest from July 2021 to June 2022. Various measurements were taken at different soil depths in July 2021 and January 2022, with continuous monthly monitoring of deep soil NO_3-N and throughfall chemistry. Note no measurements were taken in November and December 2021.

following the COVID-19 restriction phase were not included in the analysis in this study. Samples from August 2021–July 2022, except for January 2022, were also analysed for pH, colourimetrically for NH_4-N, and for DOC and total nitrogen (TN), including inorganic and organic forms of nitrogen, by C/N analyser (Shimadzu 5000, Osaka, Japan). Total inorganic nitrogen (TIN) was calculated as the sum of NH_4-N and NO_3-N.

Throughfall volume was measured at the time of sample collection. Where throughfall volume could not be measured due to overflowing containers (affecting less than 5% of measurements), a site-specific linear regression model linking throughfall to rainfall was used, separately for the winter and summer months. Rainfall data were extracted from the Met Office Haduk-Grid dataset (Hollis et al. 2019). Throughfall chemical fluxes were calculated as the product of concentration and throughfall volume, adjusted for the sampling area. Total throughfall chemical fluxes from August 2021 to July 2022 were calculated as the sum of the fluxes from all measured months.

2.2.2 | Deep Soil NO_3-N Concentrations

Based on experience from previous studies in the area, lysimeters and rhizon samplers fail to yield sufficient soil solution during the summer months in Thetford. Therefore, a cold water extraction was used to obtain soil solution NO_3-N and Cl concentrations in deep soil samples (Wada et al. 2006), which will hereafter be referred to as soil extractable NO_3-N. Aside from the three clearfell sites, soil samples were collected monthly in all sites from August 2021 until March 2022; to manage resource limitations, the number of sites was then reduced, and monitoring continued on representative sites for each stage until July 2022 ($n = 15$), but were not sampled in November and December 2021 due to COVID-19 restrictions. The three felled sites were sampled in August 2021, and then monthly after operations ceased at each site until July 2022. The soil was sampled at 60–90 cm depth (below the rooting zone) at four sampling points at each site using a Dutch Edelman auger (7 cm diameter, Eijkelkamp, Giesbeek, Netherlands). Sampling points were located adjacent to four throughfall collectors. Soil was transported in a cooler box with ice packs and then stored in a refrigerator at 4°C until analysis.

Soil samples were aggregated by site. Each month 36 g was added to 20 mL of deionised water and vortexed to suspend ions held in the soil. The soil was separated from the solution by centrifugation (Thermo Scientific Sorval ST 16R) at 4000 rpm at 4°C for 60 min. The solution was then filtered through a 0.45 μm filter and analysed for Cl and NO_3-N with a Dionex DX 500 Ion Chromatography System with AS 40 autosampler. Final Cl and NO_3-N concentrations were calculated by adjusting for soil water concentrations. Soil water concentrations were calculated using the gravimetric method, oven drying 20 g of soil at 70°C for 48 h until a constant weight was achieved and reweighing.

2.2.3 | Calculation of Leaching Fluxes

For all sites where the chloride mass balance method could be applied, water fluxes (mm) for each month were calculated as the ratio of Cl concentrations in throughfall and soil extract, multiplied by throughfall volume (mm) (e.g., Gee et al. 2005).

For each site, leaching fluxes (kg N ha^{-1}) were calculated monthly by multiplying water fluxes (mm/ha) by the concentration of NO_3-N in deep soil samples (mg/L) and presented in kg N ha^{-1}. Total leaching fluxes for the study period were calculated by aggregating monthly observations.

2.2.4 | Soil C:N Ratios, Nitrification, pH, Bulk Density and Organic Layer Depth

In July 2021, soil was sampled at four points per site aligned with the location of the throughfall collectors. At this point, five additional mature pine sites were also sampled and used to establish a baseline for sites that were expected to be felled, although only three of those sites were later felled during the timeframe of this study, giving $n = 10$ for the "mature pine" category for C:N ratios. Due to the lack of organic layer at the 0–2 year old sites, the soil was sampled by depth rather than horizon. The soil was sampled at three soil depths: (1) 0–10 cm, including the Oh layer but brushing leaf litter aside if present, (2) 10–30 cm, (3) 30–90 cm. The Ol layer, representing fresh leaf litter, was not sampled as it represents short-term, seasonal processes, whereas the interest here was in capturing differences in C:N ratios driven by variation in decomposition rates. The depth of soil was highly variable within stands. An alternative sampling point was located within the stand where the chalk bedrock was hit at less than 40 cm depth (2 sites). Soil was transported to the laboratory as described above and stored at 4°C until analysis. The samples were oven-dried for 48 h at 70°C, passed through a 5 mm sieve, mixed with a spatula, then 0.25 g was weighed per sampling point and analysed for total carbon and nitrogen using a Leco CHN elemental analyser. In February 2022, bulk density was determined at two sampling points per site, at 10–30 and 30–90 cm soil depths, by digging a pit and hammering a metal cylinder horizontally into the respective layers. The soil in each cylinder was oven-dried for 48 h at 70°C and weighed, and the bulk density was calculated by bulk density = weight of oven-dry soil in g/volume of the cylinder in cm^3. Due to limited site access, bulk density at the two sites felled in January and February was not established. An average bulk density of the mature coniferous sites studied was used instead. All of the felled sites were sampled for total C and N before felling, and grouped in the analysis of C:N ratios as mature pine. Soil carbon and nitrogen stocks at 10–30 and 30–90 cm depths were calculated from the % of C and N multiplied by the bulk density and soil layer width.

An in situ incubation described by Carter and Gregorich (2006) was used to determine the net nitrification rate. The incubation method allowed the observation of the change in NH_4-N and NO_3-N under field conditions without root uptake and deposition inputs. In July 2021, at each site, four clear PVC cylinders ($d = 5$ cm, $l = 10$ cm) with rubber caps were planted vertically at four points in the soil, with the top of the cylinder at 10 cm depth. Four small holes (< 1 cm diameter) were punched in each cap to allow gas exchange. At the time of installation, soil samples were collected at 10–30 cm depth adjacent to the cylinders and kept at 4°C until analysis. The soil in the cylinders was collected in

August 2021. All samples were analysed using the KCl extraction method. For each sampling point, 20 g of soil was used to determine soil water content by oven drying at 70°C for 48 h, and 40 g of soil (passed through 1 mm sieve) was added to 200 mL of 1 M KCl solution. The soil in the KCl solution was shaken on a horizontal shaker for 1 h, filtered and analysed within 2 days of extraction with a Skalar San^{++} continuous flow analyser. Some output values (16%) were below the limit of detection (LOD) and were considered equal to LOD/2, as recommended by the EPA (EPA 2000). NH_4 and NO_3 concentrations were adjusted for soil water content, converted to NO_3-N/NH_4-N and multiplied by bulk density and soil depth to present results in kg ha^{-1}. The increase in NO_3-N and the sum of NH_4-N concentrations during the incubation period was considered the net rate of nitrification and mineralisation, respectively (Δ kg ha^{-1}). To further explore the impacts of topsoil nitrogen transformation processes, in January 2022 additional soil samples were collected at two points at 10–30 cm depth and aggregated, and pH (H_2O) was measured in the laboratory using a pH probe. The thickness of the soil organic layer (O_F, O_H) was also measured at four sampling points per site.

2.2.5 | Statistical Analysis

2.2.5.1 | Relationships Between All Variables. R Version 4.3.1 (R Core Team 2023) was used for all statistical analysis. Using sites where NO_3-N leaching fluxes were studied for the monitoring period, a correlation matrix was created to visualise relationships between all variables (Supporting Information S3). All relationships with Pearson's r value of >0.4 were then further tested using a linear regression model. In a causality cascade, throughfall chemical fluxes were tested as predictors of soil carbon and nitrogen stocks, C:N ratios, pH, NO_3-N leaching fluxes and net rates of N mineralisation and nitrification, with pseudochronosequence stage as an additional predictor. Soil carbon and nitrogen stocks, C:N ratios, net nitrification and mineralisation, soil pH and organic layer thickness were tested as predictors of NO_3-N leaching fluxes, with pseudochronosequence stage as an additional predictor. Net nitrification and N mineralisation were also tested alongside pseudochronosequence stage as predictors of soil pH, C:N ratio and organic layer thickness. All models were tested for the parametric test assumptions by visualising the residual diagnostic plots in R. Pairwise comparisons of estimated marginal means were conducted on linear models using the emmeans package in R (Lenth 2025). Where relationships did not meet the assumptions of a linear model, variables were log transformed. Where log transformation failed to improve model fit or was not possible, a non-parametric test was used; in order to include two predictors, the response variable was rank-transformed and a linear regression model fitted between the rank-transformed response and predictors. To account for a non-uniform distribution of observed values, soil pH was additionally grouped into "low" (<6) and "high" (>7) and analysis of covariance was used to test for differences in response variables between the two groups, including pseudochronosequence stage as an additional predictor. Comparison of fluxes and soil parameters between pseudochronosequence stages.

Total water, NO_3-N leaching and throughfall chemistry fluxes for the study period were calculated by aggregating monthly observations. Total water fluxes, NO_3-N leaching and throughfall chemistry fluxes were then compared between each pseudochronosequence stage using ANOVA (R Core Team 2023). The mean deep soil extractable NO_3-N concentrations for each site across the entire study period were calculated, and these means were compared between each pseudochronosequence stage using ANOVA.

Mean C:N ratios and C and N stocks for each depth were calculated for each site, and then a mean for each stage was determined. Net nitrification rates, mineralisation and O-layer thickness were determined for each site, and then a mean for each pseudochronosequence "stage" was calculated. The difference in C:N ratios, C and N stocks, net rate of nitrification and mineralisation, O-layer thickness, soil pH and bulk density between stages was then compared using ANOVA with a significance threshold of $p<0.05$, looking for the overall significance of "stage" as an explanatory variable and using Tukey's Honest Significant Differences to conduct pairwise comparisons on significant variables.

All models were tested for the assumptions for a parametric test by the visualisation of residual diagnostic plots in R. Heterogeneity of variances was also tested using a Levene's test. Datasets not meeting test assumptions were log-transformed (net nitrification and mineralisation, deep soil water flux, deep soil NO_3-N concentrations, throughfall NO_3-N, NH_4-N, TN and DOC fluxes). Where log-transformation failed to improve model fit (organic layer thickness, C:N ratio at 0–10 cm depth), a non-parametric test (Kruskali–Wallis) was used, in which case a Dunn's test was used for pairwise comparison.

3 | Results

3.1 | Throughfall Chemistry

There was an overall significant difference in the total throughfall NO_3-N fluxes (Table 1, ANOVA, $F=2.85$, $p<0.05$), with pairwise comparison identifying stands planted 10–13 years ago as having significantly lower NO_3-N fluxes than mature broadleaf and mature pine ($t=-3.49$, $p<0.05$ and $t=-4.01$, $p<0.05$ respectively). Throughfall NH_4-N fluxes ($F=2.97$, $p>0.05$), throughfall TIN ($F=0.17$, $p>0.05$), throughfall TN fluxes ($F=0.20$, $p>0.05$) and throughfall DOC fluxes ($F=3.87$, $p=0.056$) did not display significant differences between conversion stages. Though not significantly different, the total throughfall TIN and DOC was highest in the mature pine stage, and throughfall NH_4-N was greater in the younger broadleaf stages. However, there were differences in seasonal patterns between stages in throughfall chemical fluxes (Figure 3). Sites planted with broadleaves 5–8 and 10–13 years ago had higher levels of throughfall NH_4-N fluxes in August and September 2021. Throughfall NO_3-N fluxes peaked in July in mature broadleaves. Sites planted 10–13 years ago had the greatest peak in throughfall DOC in September 2021, coinciding with a peak in NH_4-N. There was a moderate correlation between throughfall nitrogen inputs and DOC, being strongest between throughfall DOC and TIN ($r=0.60$) > TN ($r=0.58$) > NO_3-N ($r=0.48$) > NH_4-N ($r=0.30$).

All stages leached less NO_3-N compared to the total input of inorganic nitrogen via throughfall (Table 1), with the difference

TABLE 1 | The sum total NO_3-N leaching fluxes deep soil drainage, throughfall chemical fluxes and mean soil extractable NO_3-N concentrations from stands at different stages in the conifer-to-broadleaf conversion process, between August 2021 and July 2022.

	Mature pine	0–2 years	5–8 years	10–13 years	Mature broadleaf	p
Total throughfall volume (mm)	156.63	—	138.14	149.25	131.90	0.82
Total throughfall NO_3-N (kg N/ha)	21.64 (6.20)	—	10.40 (2.00)	8.56 (5.11)	21.29 (5.26)	0.02
Total throughfall NH_4-N (kg N/ha)	8.61 (7.00)	—	14.57 (2.56)	20.75 (13.14)	3.90 (2.58)	0.09
Total throughfall TIN (kg N/ha)	30.25 (11.39)	—	24.97 (1.13)	29.31 (18.25)	25.18 (3.26)	0.9
Total throughfall TN (kg N/ha)	51.86 (41.19)	—	34.36 (0.95)	38.27 (24.60)	27.10 (4.63)	0.8
Total throughfall DOC (kg N/ha)	94.58 (40.75)	—	45.62 (4.11)	59.15 (15.10)	47.69 (5.00)	0.06
Total drainage (mm)	145.12 (59.51)	—	100.34 (54.35)	130.91 (15.54)	92.26 (21.83)	0.64
Mean soil extractable NO_3-N (mg N/L)	30.88 (13.90)	23.64 (9.61)	15.34 (6.40)	19.4 (8.57)	12.46 (3.97)	0.18
Total NO_3-N leaching (kg N/ha)	23.81 (1.85)	—	16.11 (8.61)	14.53 (1.64)	10.72 (2.01)	0.10

Note: All variables were not measured November–January. +/− Standard Error is presented in brackets, $n = 3$. The p-value represents the significance of each overall model.

between TIN and NO_3-N leaching smallest in mature pine (6.44 kg ha^{-1}) and greatest in the broadleaves planted 10–13 years ago (14.78 kg ha^{-1}).

3.2 | Patterns in NO_3-N Leaching Fluxes, Water Fluxes and Mean NO_3-N Concentrations

There was no overall significant difference in the total observed deep soil water flux between conversion stages (Table 1, ANOVA, $F = 0.59$, $p > 0.05$). Mean deep soil extractable NO_3-N concentrations for the whole study period were not significantly different between stages (Table 1, ANOVA, $F = 2.28$, $p > 0.05$). Similarly, there was no significant difference between conversion stages in the total observed NO_3-N leaching for the whole monitoring period (Table 1, ANOVA, $F = 2.87$, $p > 0.05$). The three mature conifer sites studied for the year had total NO_3-N leaching fluxes of 26.01, 23.92 and 21.49 kg N ha^{-1}. The three mature broadleaves studied for the full monitoring period had total NO_3-N leaching fluxes of 13.56, 9.40 and 9.20 kg N ha^{-1}.

There were differences in seasonal patterns between conversion stages in NO_3-N leaching fluxes, deep soil drainage and deep soil extractable NO_3-N concentrations (Figure 4). The highest peak in deep soil extractable NO_3-N concentrations and NO_3-N leaching flux was observed under mature Pine. Deep soil extractable NO_3-N concentration in mature broadleaf stands increased from May to July 2022, whereas concentrations remained stable in all other sites.

3.3 | Soil C and N Stocks and Cycling

The net rates of nitrification and mineralisation were not significantly different between the conversion stages (Figure 5, ANOVA, $F = 1.93$, $p > 0.05$ and $F = 1.27$, $p > 0.05$) or species (ANOVA, $F = 3.42$, $p > 0.05$ and $F = 2.19$, $p > 0.05$). The mean net rate of nitrification was positive in the mature pine, 0–2 year old and 5–8 year old sites, but negative in 10–13 year old and mature broadleaf sites, being below the overall mean (red line, Figure 5a). There was a wide variation in N mineralisation across the mature pine sites, but it was generally low in broadleaved sites, with all the broadleaved stages being below the overall mean across sites (Figure 5b). The organic layer thickness and soil pH value at 10–30 cm soil depth were also not significantly different between the stages (Figure 5c, Kruskal–Wallis, $X^2 = 8.71$, $p > 0.05$ and Figure 5d, ANOVA, $F = 1.58$, $p > 0.05$). Sites planted 0–2 years ago had no organic layer present, and there was little variation in the thickness of the organic layer between mature pine stands. The soil pH in sites planted 10–13 years ago was higher than the other stages, being above the overall mean, though not significantly so.

The C:N ratio at soil depth 0–10 cm significantly differed between forest conversion stages (Figure 6a, Kruskal–Wallis, $X^2 = 43.98$, $p < 0.01$). The 0–10 cm C:N ratio at sites planted 0–2 years ago (C:N = 12.1) was significantly lower than in mature pine (C:N = 18.9, Dunn's test, $p < 0.01$) and mature broadleaved sites (C:N = 15.6, Dunn's test, $p < 0.05$). Broadleaf sites planted 5–8 years ago (C:N = 14.3) and sites planted 10–13 years ago (C:N = 13.6) were significantly lower than that in mature pine stands (Dunn's test, $p < 0.01$), but were not significantly different to each other.

There was a significant difference in the C:N ratio at soil depths 10–30 cm (Figure 6b, Kruskal–Wallis, $X^2 = 19.41$, $p < 0.01$), with sites aged 0–2 years having significantly lower C:N ratios than mature broadleaf (Dunn's test, $p < 0.05$) and broadleaf sites planted 10–13 years ago (Dunn's test, $p < 0.05$). There were no significant differences in C:N ratios at 30–90 cm depth between the stages (Figure 6c, ANOVA, $F = 0.44$, $p > 0.05$), nor the total C (Supporting Information

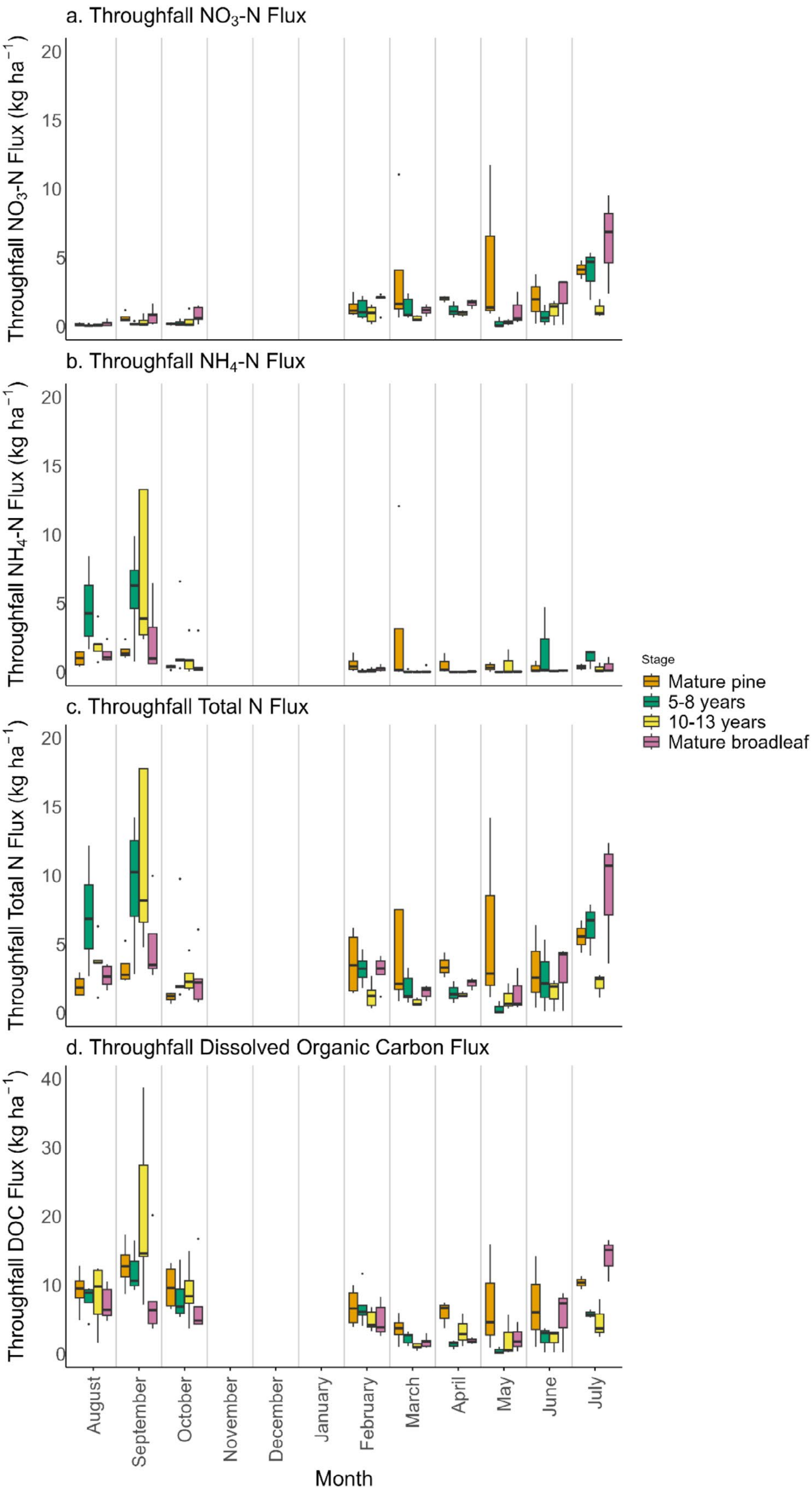

FIGURE 3 | Legend on next page.

FIGURE 3 | Mean (a) throughfall NO_3-N fluxes (kg ha^{-1} month^{-1}), (b) throughfall NH_4-N fluxes (kg ha^{-1} month^{-1}), (c) throughfall total N fluxes (kg ha^{-1} month^{-1}) and (d) throughfall DOC fluxes (kg ha^{-1} month^{-1}), between August 2021 and July 2022, in stands at different stages in the conifer-to-broadleaf conversion process: (1) Mature pine (n = 5 until March 2022, then n = 3 until July 2022), (2) 5–8 years post-conversion (n = 5 until March 2022, then n = 3 until July 2022), (3) 10–13 years post-conversion (n = 5 until March 2022, then n = 3 until July 2022), (4) mature broadleaf (n = 5 until March 2022, then n = 3 until July 2022).

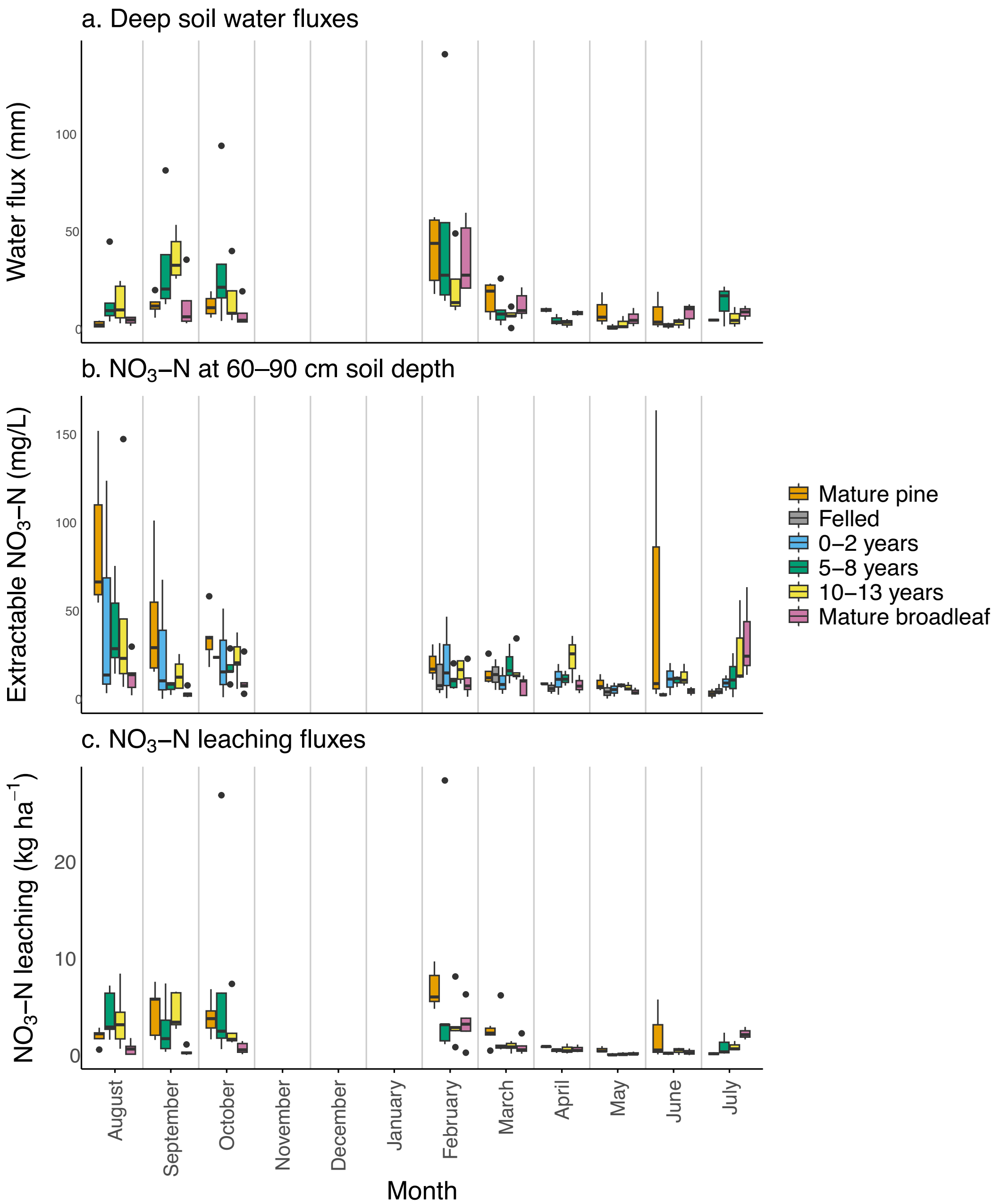

FIGURE 4 | (a) Total deep soil drainage fluxes (mm), (b) mean deep soil extractable NO_3-N concentrations (mg L^{-1}) and (c) total NO_3-N leaching fluxes (kg N ha^{-1} month^{-1}), between August 2021 and July 2022, in stands at different stages in the conifer-to-broadleaf conversion process: (1) Mature pine (n = 5 until March 2022, then n = 3 until July 2022), (2) Felled (n = 1 from September 2021, then n = 3 from February 2022–July 2022), (3) 0–2 years post-conversion (n = 3 until March 2022, then n = 2 until July 2022), (3) 5–8 years post-conversion (n = 5 until March 2022, then n = 3 until July 2022), (4) 10–13 years post-conversion (n = 5 until March 2022, then n = 3 until July 2022), (5) mature broadleaf (n = 5 until March 2022, then n = 3 until July 2022).

S4, ANOVA, $F = 0.88$, $p > 0.05$ respectively) and total N stocks (Supporting Information S4, ANOVA, $F = 1.03$, $p > 0.05$ respectively) at 30–90 cm. In soils at 30–90 cm depth, there was an increase in N stocks 10–13 years after replanting sites with broadleaved species, although the increase is not quite significant ($p = 0.07$).

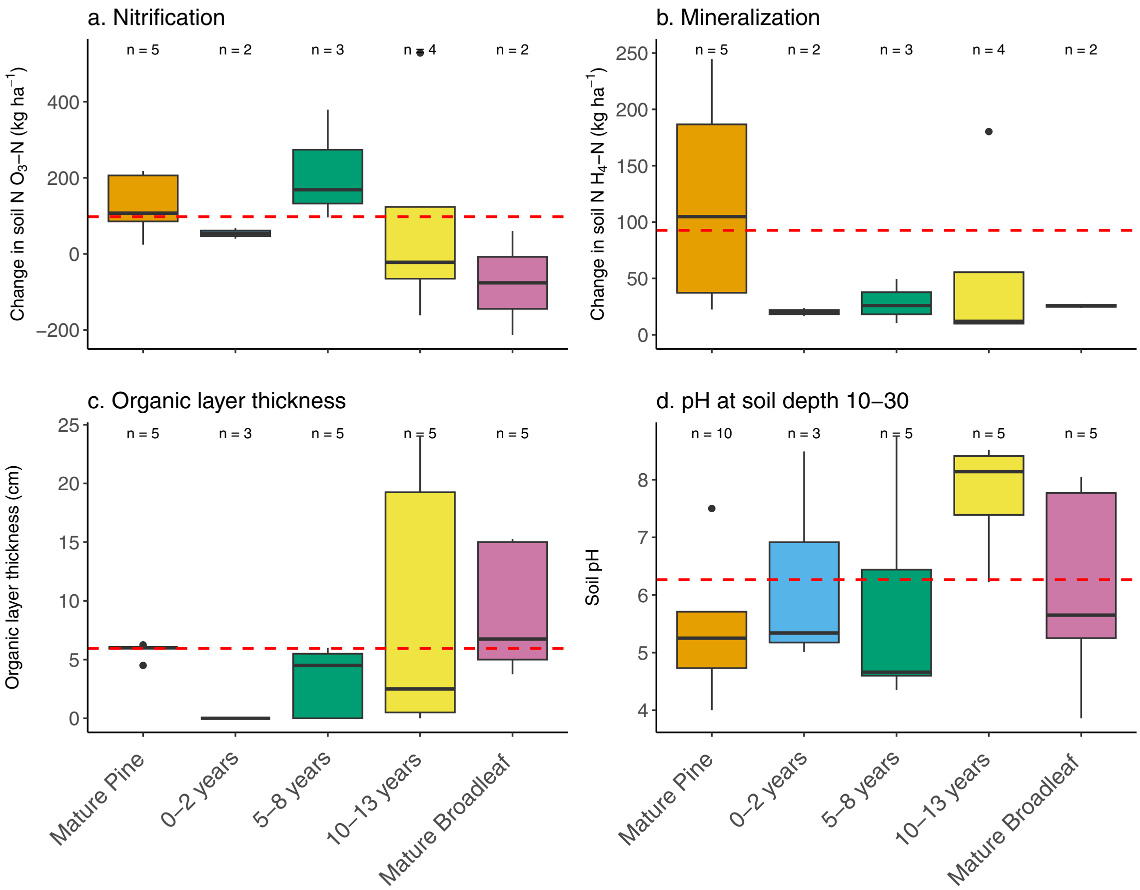

FIGURE 5 | The (a) rate of change in NO_3-N extracted by the KCl extraction method at 10–20 cm soil depth between July and August 2021 (in situ net nitrification), (b) the sum of NH_4-N extracted by the KCl extraction method at 10–20 cm soil depth in July and August 2021 (in situ mineralisation), (c) organic layer thickness (cm) and (d) pH(H_2O) of soil at depth 10–30 cm, in stands at different stages in the conversion process: (1) mature pine, (2) 0–2 years post-conversion, (3) 5–8 years post-conversion, (4) 10–13 years post-conversion and (5) mature broadleaf. The dashed red line indicates the overall mean across all groups.

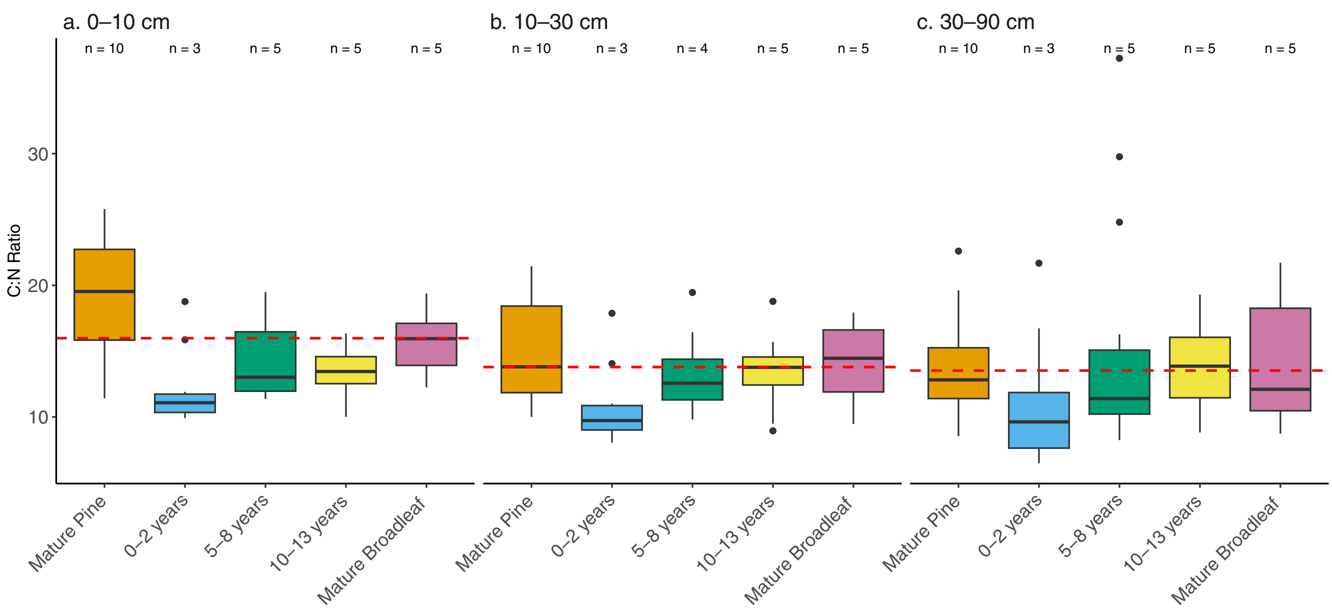

FIGURE 6 | The C:N ratio of soil at (a) 0–10 cm, (b) 10–30 and (c) 30–90 cm depth in mature pine ($n = 10$), former pine sites replanted with broadleaved species: 0–2 years ago ($n = 3$), 5–8 years ago ($n = 5$) and 10–13 years ago ($n = 5$) and mature broadleaved sites ($n = 5$). The dashed red line indicates the overall mean for each depth across all groups.

TABLE 2 | The relationships between throughfall chemistry and soil nitrogen cycling processes observed across a pseudochronosequence of sites in the conifer-to-broadleaf conversion process.

Predictor	Response	Standardised β-coefficient	95% confidence intervals		p-value	+ Stage as a predictor	
			Lower bound	Upper bound		Adj-R^2	p-value
Throughfall pH	Net nitrification rate	0.9	−1.3	3.1	0.23	0.05	0.44
Throughfall NO3-N flux	Net nitrification rate	0.52	−1.77	2.81	0.46	−0.16	0.63
Throughfall NH4-N flux	Net nitrification rate	−0.58	−1.99	0.82	0.08	0.33	0.22
Throughfall TIN flux	Net nitrification rate	−0.1	−1.36	1.16	0.095	0.29	0.25
Throughfall TN flux	Net nitrification rate	−0.17	−1.38	1.03	0.07	0.36	0.20
Throughfall DOC flux	Net nitrification rate	−0.04	−1.59	1.51	0.35	−0.08	0.56
Soil C:N 0–10 cm	Net nitrification rate	−0.54	−3.00	1.93	0.48	−0.17	0.64
Soil C:N 10–30 cm	Net nitrification rate	0.02	−1.30	1.34	0.13	0.21	0.31
Soil C:N 30–90 cm	Net nitrification rate	0.63	−0.24	1.50	**0.008**	0.72	**0.03**
Organic layer thickness	Rank-transformed net nitrification rate	0.68	−1.11	0.46	**0.01**	0.73	**0.03**
Throughfall pH	NO3-N Leaching Flux	0.85	−2.09	0.22	0.06	0.56	**0.04**
Throughfall NO3-N flux	NO3-N Leaching Flux	−0.34	−1.44	0.76	0.48	0.30	0.18
Throughfall NH4-N flux	NO3-N Leaching Flux	0.07	−0.73	0.87	0.84	0.25	0.22
Throughfall TIN flux	NO3-N Leaching Flux	−0.02	−0.68	0.63	0.94	0.24	0.22
Throughfall TN flux	NO3-N Leaching Flux	0.01	−0.65	0.67	0.97	0.24	0.22
Throughfall DOC flux	NO3-N Leaching Flux	0.16	−0.72	1.04	0.68	0.26	0.20
Soil C:N 0–10 cm	NO3-N Leaching Flux	−0.63	−1.60	0.34	0.17	0.43	0.09
Soil C:N 10–30 cm	NO3-N Leaching Flux	0.08	−0.20	0.36	0.50	0.87	**0.002**
Soil C:N 30–90 cm	NO3-N Leaching Flux	0.66	0.32	0.99	**0.003**	0.81	**0.003**
Soil pH 10–30 cm	NO3-N Leaching Flux	0.63	0.10	1.17	**0.03**	0.64	**0.03**
Organic layer thickness	NO3-N Leaching Flux	−0.04	−0.70	0.61	0.89	0.25	0.22
Log-transformed net nitrification rate	NO3-N Leaching Flux	0.52	−1.57	2.61	0.38	0.94	**0.04**

Note: The standardised β-coefficient gives the effect sizes for pairwise models between each predictor and response variable, with corresponding 95% confidence intervals and p-values. Also reported are the adjusted R^2 values and overall model p-values for regressions that included pseudochronosequence stage as an additional predictor. Bold values indicate statistical significance at $p < 0.05$.

3.4 | Relationships Between Soil C, N, pH, Net Nitrification, Throughfall Chemistry and Deep Soil NO_3-N Leaching Fluxes

When stage was included as a predictor, net nitrification rate was not significantly related to average throughfall pH (linear regression model, $F = 1.12$, Table 2), total throughfall NO_3-N (linear regression model, $F = 0.69$, Table 2) or throughfall DOC flux (linear regression model, $F = 0.83$, Table 2). Though the overall regression model was significant when including stage alongside throughfall DOC as predictors of soil C:N ratios at depth 0–10 cm ($F = 4.03$, $p = 0.05$, $R^2 = 0.52$, Supporting Information S5), throughfall DOC fluxes were not significantly related to C:N ratios (linear regression model, $t = -0.22$, $p > 0.05$, Supporting Information S5), nor were deep soil C:N ratios at 10–30 and 30–90 cm (linear regression model, $t = 0.32$ and 0.02, $p > 0.05$, Supporting Information S5). Though the overall regression model was significant when testing throughfall NO_3-N fluxes and stage as predictors of soil C:N ratios at 0–10 cm depth (linear regression model, $F = 4.34$, $R^2 = 0.55$, $p < 0.05$, Supporting Information S5), throughfall NO_3-N fluxes were not significantly related to C:N ratio at 0–10 cm (linear regression model, $t = 0.66$, $p > 0.05$, Supporting Information S5) and there were no significant differences between stages ($p > 0.05$). Including stage as a predictor, soil C:N ratios at 10–30 and 30–90 cm depth were not significantly correlated with throughfall NO_3_N (linear regression model, $F = 0.04$ and 0.39, $R^2 = -0.62$ and −0.28, $p > 0.05$, Supporting Information S5), throughfall NH_4-N (linear regression model, $F = 0.37$ and 0.36, $R^2 = -0.61$ and −0.29, $p > 0.05$, Supporting Information

S5), or throughfall TN ($F = 0.06$ and 0.39, $R^2 = -0.60$ and -0.29, $p > 0.05$, Supporting Information S5).

Including stage as a predictor, there was no significant relationship between the net nitrification rate and C:N ratios of soil at 0–10 cm depth (linear regression model, $F = 0.67$, $R^2 = -0.17$, $p > 0.05$, Table 2), nor at 10–30 cm depth (linear regression model, $F = 1.58$, $R^2 = 0.21$, $p > 0.05$, Table 2). However, the overall regression model with stage and nitrification rate as predictors of C:N ratios of soil at 30–90 cm depth was significant (linear regression model, $F = 6.74$, $R^2 = 0.72$, $p < 0.05$, Table 2), with a significant relationship between soil C:N ratio and nitrification rate ($t = -4.27$, $p < 0.01$, Table 2), accompanied by a large standardised effect size (Standardised β-coefficient = 0.63). Soil pH values ranged from 4.35 to 8.52, with no sites represented between pH 6 and 7; therefore, pH was grouped into low (≤6) and high (≥7) classes for analysis. Controlling for pseudochronosequence stage, net nitrification rates were significantly greater in the low-pH group than in the high-pH group (ANCOVA, $F = 10.28$, $p < 0.05$). Including stage as a predictor, there was a significant relationship between organic layer thickness and rank-transformed nitrification (Table 2, linear regression model, $t = 3.92$, $p < 0.01$), with the model explaining 73% of variation in nitrification (Table 2, linear regression model, $F = 6.97$, $p < 0.05$). Though the overall regression model including net nitrification rate and stage as predictors of rank-transformed NO_3-N leaching fluxes was significant (linear regression model, $F = 5.70$, $R^2 = 0.68$, $p < 0.05$), net nitrification rate was not significantly related to rank-transformed NO_3-N leaching fluxes (Table 2, linear regression model, $t = 0.33$, $p > 0.05$). The mineralisation rate was not significantly related to soil pH (ANCOVA, $t = -1.43$, $p > 0.05$) or C:N ratio ($t = -0.1$, $p > 0.05$) at 10–30 cm soil depth, but was related to organic layer thickness (linear regression model, $t = 3.06$, $p < 0.05$) when stage was included as a predictor though the overall model was not significant ($F = 2.76$, $R^2 = 0.44$, $p > 0.05$).

In the regression model including pseudochronosequence stage as an additional predictor, deep soil NO_3-N fluxes were not significantly related to log-transformed net nitrification rates ($t = 1.13$, $p > 0.05$), but pairwise comparison suggested the overall significance of the regression model (Table 2, $p < 0.05$) was explained by a marginal difference in NO_3-N leaching fluxes between the mature pine and sites planted 5–8 years ago (emmeans, $t = -6.96$, $p = 0.05$). NO_3-N leaching fluxes did not show a consistent relationship with soil pH at 10–30 cm soil depth across the pseudochronosequence (ANCOVA, $F = 0.21$, $p > 0.05$). NO_3-N leaching fluxes were not significantly related to soil C:N ratios at 0–10 cm depth when stage was included as a predictor (Table 2, linear regression model, $t = 0.15$, $p > 0.05$), but were significantly related to C:N ratio at 30–90 cm (linear regression model, $t = 0.7$, $p < 0.01$), with a large standardised effect size and narrow confidence intervals (Standardised β-coefficient = 0.66, 95% CI = 0.32 and 0.99). When pseudochronosequence stage was included as a predictor alongside C:N ratios at 10–30 cm depth, the overall regression model was significant (Table 2, $F = 17.91$, $p < 0.04$), although C:N ratios at 10–30 cm depth were not a significant predictor of NO_3-N leaching fluxes (linear regression model, $t = 0.23$, $p > 0.05$). Post hoc pairwise comparisons suggested that this model significance was associated with differences among pseudochronosequence stages, with significant contrasts between mature pine stands and sites planted 5–8 years ago ($t = -6.83$, $p < 0.05$), 10–13 years ago ($t = -5.24$, $p < 0.05$) and mature broadleaf ($t = -7.35$, $p < 0.01$). Pairwise comparison also identified significant differences in the relationship between NO_3-N leaching fluxes and C:N ratio at 30–90 cm depth between mature pine and sites planted 5–8 years ago ($t = -4.69$, $p < 0.01$), 10–13 years ago ($t = -4.72$, $p < 0.01$) and mature broadleaf ($t = -6.00$, $p < 0.01$).

There was no significant relationship between NO_3-N leaching fluxes and throughfall DOC fluxes when stage was included as a predictor (Supporting Information S5, linear regression model, $t = 0.43$, $p > 0.05$). There was also no significant relationship between NO_3-N leaching fluxes and throughfall NO_3-N fluxes (Supporting Information S5, linear regression model, $t = -0.74$, $p > 0.05$), throughfall NH_4-N fluxes (linear regression model, $t = 0.2$, $p > 0.05$), throughfall TN fluxes (linear regression model, $t = 0.04$, $p > 0.05$) and throughfall TIN fluxes (linear regression model, $t = -0.08$, $p > 0.05$). There was no significant interaction between the response variables, species or stage and throughfall total TIN fluxes, as predictors of NO_3-N leaching ($F = 0.72$ and 0.78 respectively, $p > 0.05$).

4 | Discussion

In this study, NO_3-N leaching was on average 10.7 kg N ha^{-1} from mature broadleaved stands and 23.8 kg N ha^{-1} from conifers total across the observed months. A leaching flux of 23.8 kg N ha^{-1} over eight months is high for European forests where leaching rarely exceeds 20 kg N ha^{-1} year^{-1} (Macdonald et al. 2002; Dise and Wright 1995). The present study also missed observations during the winter months where soil drainage fluxes may have peaked and driven higher NO_3-N leaching fluxes, so the totals here underestimate the full annual sum of NO_3-N leaching fluxes. The highest observed leaching flux occured near major roads or human habitation. On average, NO_3-N in deep soil solution extract exceeded the Water Framework Directive threshold value of 11 mg N L^{-1} in all stages, especially mature conifers (30.9 mg N L^{-1}) and 0–2 year-old broadleaves (23.6 mg N L^{-1}). It is worth noting that whilst there were no significant differences in NO_3-N leaching between stages, leaching from broadleaved stands was generally lower than the conifers, but variation likely driven by site-specific factors, such as edge effects, may have masked significant differences.

Our observation period was relatively dry in eastern England (UKCEH et al. 2022). Summer and winter air temperature was near normal in 2021, but rainfall was below average in August and particularly dry in November (Kendon et al. 2022). Rainfall in the region was less than two-thirds normal rainfall in 2022, with conditions particularly dry January–August 2022, accompanied by higher than average daily maximum air temperature (Kendon et al. 2023). There was a severe soil moisture drought in Southern England in the latter part of our observation period, with declining soil moisture from April 2022 (Barker et al. 2024). A more extended study period covering eventual wetter periods may have captured even greater peaks in NO_3-N leaching; Leitner et al. (2020) observed peaks in NO_3-N leaching fluxes occurring following drought periods. Considering this present study observed relatively high NO_3-N leaching fluxes for forest ecosystems despite likely not capturing the maximum peaks in NO_3-N leaching, the need for forest management practices to reduce NO_3-N leaching from conifer plantations is evident.

4.1 | Hypothesis 1: *Deep Soil NO_3-N Leaching Fluxes Will Remain Elevated in the First Decade After Planting Broadleaves on Former Coniferous Stands*

Hypothesis 1 was formulated on the understanding that tree species affects organic matter inputs, microclimatic variables and soil chemical properties, leading to the suggestion that converting conifer sites to broadleaved species would stimulate decomposition and nitrogen transformation processes. Consistent with this expectation, the reduced soil C:N ratios at 0–10 cm depth following conifer-to-broadleaf conversion in this study are indicative of active decomposition and loss of organic matter (Eastman et al. 2021). It also suggests a change in organic matter inputs; previous studies suggested tree-species differences in forest floor and topsoil C:N ratios were driven by the lignin and N content of litter (Cools et al. 2014; Hobbie et al. 2006), and are related to nitrogen-transforming microbial community composition (Ribbons et al. 2018). As lower C:N ratios have been associated with higher levels of NO_3-N leaching fluxes in previous studies (e.g., Dise et al. 1998), it was expected that NO_3-N leaching would remain elevated for at least a decade after replanting, but this was not observed. The disconnect between surface and deep soil processes may be reflective of the variation in nitrogen transformation processes caused by site-specific factors, such as edge effects from adjacent farmland which can influence NO_3-N fluxes due to attenuation of nitrogen inputs with distance from edge (Vanguelova and Pitman 2019). Nitrogen saturation was widespread across sites; the highest C:N ratios observed were at 0–10 cm soil depth where the average was below < 20 across groups. This nitrogen saturation has masked any species- or management-related effects on NO_3-N leaching, and studies under N-limited conditions could be insightful. While C:N ratios at 0–10 cm soil depth were significantly lower in all the observed stages post-conversion, there could be a time-lag in the response of ammonia-oxidising community composition to conversion, decoupling NO_3-N leaching fluxes from upper soil C:N ratios. Hypothesis 1 was not supported by observation but the change in C:N ratios suggests further avenues to explore under this scenario.

4.2 | Hypothesis 2: *Topsoil Nitrification Rates Are Increased Following Conifer-To-Broadleaf Conversion, Driving an Increase in NO_3-N Leaching and a Decrease in Topsoil pH*

Net nitrification rates observed in this study were consistent with those reported by Ambus et al. (2006), confirming the reliability of the in situ incubation method used here. The negative net nitrification values in the 10–13 year old broadleaf and mature broadleaved stands indicate greater immobilisation or gaseous nitrogen losses. N_2O emissions are typically higher from deciduous soils; Ambus et al. (2006) reported N_2O emissions from deciduous soils were 13 ng N_2O-N cm^{-3} d^{-1} compared to 4 in coniferous soils ng N_2O-N cm^{-3} d^{-1}. Sites with higher soil pH at 10–30 cm depth showed lower or negative net nitrification rates than those with low pH, contrary to expectation. While some studies found increasing soil pH boosted nitrification (DeForest and Otuya 2020; Ste-Marie and Pàre 1999), this effect depends on microbial community composition. Acid-adapted *Nitrosospira* spp. can dominate over *Nitrosomonas* spp. at lower soil pH (Killham 1994), which was observed in Thetford Forest previously (Thorpe 2011), and lower soil pH perhaps facilitated greater ammonia-oxidising activity by this group. However, soil pH at 10–30 cm soil depth was not affected by pseudochronosequence stage. The dominance of underlying site geology may have overridden the expected microbial and tree species-control on soil pH, limiting a change in net nitrification rates after conifer-to-broadleaf conversion.

The absence of a relationship between net nitrification rates and NO_3-N leaching fluxes is challenging to explain. Whilst nitrification was correlated with C:N ratios in the deepest soil layers, and C:N ratios at 30–90 cm depth were correlated with NO_3-N leaching fluxes, this did not translate into a significant relationship between net nitrification rates and NO_3-N leaching fluxes. However, only summer nitrification rates were measured here; it is still possible that autumn nitrification rates, when broadleaf litterfall inputs peak, would display a relationship with NO_3-N leaching fluxes (Aubert et al. 2005). A longer study duration could reveal connections between upper soil and deep soil processes.

4.3 | Hypothesis 3: *Differences in Throughfall Chemistry (Dissolved Organic Carbon, Inorganic Nitrogen) Between Forest Management Stages Will Explain Variation in Soil Nitrification Rates, Soil pH and NO_3-N Leaching*

There was a significant difference in throughfall NO_3-N fluxes between the forest management stages in the present study. Throughfall NO_3-N fluxes were higher than NH_4-N fluxes in mature broadleaf and pine stands, whereas the opposite was true for the two younger broadleaf stages, which may reflect a smaller effect of canopy nitrification in younger stands. Whilst a significant difference in throughfall DOC could be expected between stands with different species (Verstraeten et al. 2018, 2023), this was not observed in the present study. The broadleaf stands could have been influenced by pollen from surrounding conifer stands and all sites may have experienced variation in local-scale inputs such as crop pollen, plant debris and possibly local atmospheric inputs from anthropogenic activities. Throughfall pH, DOC, TN, NH_4-N, NO_3-N and TIN did not display a significant relationship with NO_3-N leaching fluxes, nitrification rates or soil pH. TIN deposition is a known major driver of NO_3-N leaching fluxes across Europe (Van der Salm et al. 2007; Macdonald et al. 2002; Dise and Wright 1995), with some studies finding this relationship is not dependent on forest type (e.g., Van der Salm et al. 2007; Macdonald et al. 2002). The absence of a significant relationship in this study is therefore unexpected, but suggests NO_3-N leaching is modulated by internal soil processes, or variation between sites was not large enough to cause differences in NO_3-N leaching. Site characteristics may also have driven soil pH, which was linked to nitrification rates, rather than deposition inputs.

4.4 | Implications for Forest Management

The leaching fluxes in mature conifer stands ranged from 21.49 to 26.01 kg N ha^{-1} total over the 8 months observed. This remains below the average leaching of NO_3-N from UK agricultural land, which ranges between 39 and 88 kg N ha^{-1} year^{-1} (Silgram et al. 2001), but it is high for a forest ecosystem. Leaching fluxes from the mature broadleaves studied for the 8 months had lower total NO_3-N leaching fluxes, ranging from 9.20 to 13.56 kg N ha^{-1}, supporting the expectation that mature broadleaf forest could be more favorable for water quality. However, on average, leaching fluxes from the sites planted with broadleaves 5–8 and 10–13 years ago still remained between the two mature stages, highlighting the time it may take to fully realise the benefits of transitioning systems. This study on nitrogen-saturated soils found evidence of changes in C:N ratios at 0–10 cm soil depth following conifer-to-broadleaf conversion, which indicates the potential for the conversion process to stimulate decomposition and nitrogen transformation processes, even if clear relationships with nitrification rates and NO_3-N fluxes were not observable here due to overriding local-scale drivers. The understanding of the impacts of converting conifer plantations to broadleaf forest would benefit from dedicated long-term experiments, exploring interactions with initial nitrogen saturation status, soil type and edge effects.

5 | Conclusion

Our findings suggest that changes in forest management did not have clear impacts on soil nitrogen cycling in the study area. Soil pH emerged as a key driver of nitrification, which impacted deep soil C:N ratios. Local-scale variation likely masked some effects, highlighting the need to understand interactive effects between forest management interventions and site characteristics to assess the capacity for interventions to improve water quality. Though site-dependent, the young and mature broadleaf sites did generally display lower levels of NO_3-N leaching than the mature conifers, and planting broadleaves on former conifer sites may eventually yield benefits to water quality in some cases.

Author Contributions

Caitlin Lewis: conceptualization, investigation, writing – original draft, methodology, validation, visualization, writing – review and editing, data curation, project administration, formal analysis. **Elena Vanguelova:** conceptualization, methodology, supervision, data curation, project administration, resources, writing – review and editing. **Matthew Ascott:** resources, supervision, project administration, writing – review and editing, conceptualization, methodology, funding acquisition. **Lucie Jerabkova:** conceptualization, methodology, writing – review and editing, resources. **Joshua Deakins:** methodology, data curation, writing –review and editing. **Martin Lukac:** conceptualization, methodology, investigation, funding acquisition, writing – review and editing, project administration, supervision, resources.

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Data Availability Statement

The data that support the findings of this study are openly available in Zenodo at https://zenodo.org/records/15325792.

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Supporting Information

Additional supporting information can be found online in the Supporting Information section. **Supporting Information S1.** Groundwater abstraction in the Little Ouse and Thet catchment. *Source:* Environment Agency. **Supporting Information S2:** Details of the field study sites sampled in Thetford Forest, UK, between August 2021 and 2022 to measure variables related to N cycling processes. **Supporting Information S3:** The correlation matrix created to identify relationships between variables measured in forest stands across Thetford Forest. Forest stands studied included mature broadleaf and coniferous species and sites where broadleaves were planted on conifer clearfells 5–8 and 10–13 years prior to the start of the study. **Supporting Information S4:** In stands at different stages in the conifer-to-broadleaf conversion process, the Soil nitrogen stocks and carbon stocks at depths 10–30 and 30–90 cm. **Supporting Information S5:** The relationships between throughfall chemistry and soil nitrogen cycling processes observed across a pseudochronosequence of sites in the conifer-to-broadleaf conversion process. The standardised β-coefficient gives the effect sizes for pairwise models between each predictor and response variable, with corresponding 95% confidence intervals and p-values. Also reported are the adjusted R^2 values and overall model p-values for regressions that included pseudochronosequence stage as an additional predictor. Bold values indicate statistical significance at $p < 0.05$.