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Research article

Water quality trends in river catchments draining to the Great Barrier Reef: an analysis of ~40 years of data

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ABSTRACT

Analysis of water quality trends in river catchments provides critical insights into the influence of climate, landscape/land use change and management drivers thereby informing evidence-based policy interventions. Long-term water quality datasets from nine monitoring sites across the Great Barrier Reef (GBR) catchment of northeastern Australia were carefully curated and analysed using Generalised Additive Mixed Models (GAMMs) to identify both long-term (~linear) and episodic (non-linear) trends in dissolved inorganic nitrogen (DIN) and suspended sediment (SS) concentrations. The GAMMs approach addresses common shortfalls in long-term time series including temporal autocorrelation and data gaps. Most sites exhibited episodic behaviour, with periods of increasing and decreasing concentrations over monitoring periods ranging from 18 to 37 years. For DIN, one site showed a significant linear increase in DIN concentrations, while seven sites displayed significant linear declines. These trends largely align with changes in nitrogen fertiliser application, although interpretation is complicated in catchments with mixed upstream land uses. For SS, seven sites exhibited significant downward linear trends, and one site showed increasing concentrations reflecting the influence of multiple drivers including land use changes, climate cycles, variable ground cover and management interventions. Although the underlying causes of these changes are not yet fully resolved, the observed trends are nevertheless encouraging. Further work is needed to evaluate how constituent loads delivered to the GBR may be changing under current and projected climate conditions.

1. Introduction

Globally, riverine concentrations of nutrients and sediment have changed markedly due to catchment modifications for agriculture, mining, urban development, and the construction of water infrastructure on streams, such as dams and drainage networks (Vorosmarty et al., 2003; Wang et al., 2011; Galloway et al., 2004; Syvitski et al., 2005). Profound negative impacts on receiving environments have been observed in regions where substantial changes in nutrient and sediment fluxes have occurred. For example, increased riverine nutrient fluxes have contributed to the deterioration of marine environments by triggering eutrophication, creating marine ‘dead zones’, and degrading ecosystems such as coral reefs (Beman et al., 2005; Fabricius, 2005; Diaz-Pulido et al., 2024). Increased sediment fluxes have led to sedimentation of stream networks, burial of corals and seagrass meadows and reductions in water clarity (Beman et al., 2005; Fabricius, 2005; Fabricius et al., 2016; Fabricius et al., 2005; Turak et al., 2021; Bartley et al., 2014; Bainbridge et al., 2018; Collier et al., 2024; Diaz-Pulido et al., 2024; Hughes and Vadas Jr, 2021). In contrast, dramatic reductions in nutrients and sediment, often due to dams and impoundments, result in changes to the nutrient balance leading to the collapse of fisheries and wetlands as well as increased coastal erosion (e.g. Stockner

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et al., 2000; Syvitski et al., 2005; Yang et al., 2006; Zhang et al., 2022; Ibáñez et al., 2023). In response to these adverse impacts, there have been large investments to remediate landscapes and improve land management practices to address poor water quality (Eberhard et al., 2021; Commonwealth of Australia, 2023; Hawdon et al., 2025; Hughes and Vadas Jr, 2021). Long-term water quality monitoring programs conducted alongside catchment management initiatives are critical for documenting trends in riverine nutrient and sediment concentrations and for quantifying the success of remediation activities.

Where adequate riverine water quality data are available to assess temporal trends, a range of statistical methods have been applied. These approaches range from simple linear regression–based trend analyses (e.g. Walling and Fang, 2003; Horowitz, 2010) to non-parametric methods such as the Mann–Kendall and seasonal Kendall tests, which have often been adapted to deal with serial correlation effects in rivers across China and New Zealand (e.g. Rustomji et al., 2008; Snelder et al., 2021). Other studies have identified 'paired' periods within long-term water quality records that represent comparable hydrological conditions to help quantify changes in water quality due to land use change (e.g. Wang et al., 2015). In the last twenty years, approaches specifically tailored to water quality trend detection have been developed, notably the Weighted Regressions on Time, Discharge, and Season (WRTDS) method (Zhang et al., 2026). WRTDS provides estimates of the flow-normalised concentration for each day within a record, reducing the influence of inter-annual streamflow variability and thereby enabling clearer identification of long-term, systematic changes in water quality. This approach has been used widely to explore trends in water quality internationally (e.g. Hirsch et al., 2010; Murphy and Sprague, 2019; Zhang et al., 2021) and more recently in Australia (e.g. Guo et al., 2025). However, for river systems characterised by highly variable flow regimes, statistical methods are required that are sufficiently flexible to represent complex, non-linear relationships between constituent concentration and streamflow.

Generalised additive models (GAMs) are flexible statistical models that combine both parametric and semi-parametric components allowing for the modelling of underlying seasonal patterns in data and complex, non-linear changes over time (Wood, 2017). They have been widely used to estimate non-linear trends in water quality and flow measurements, including in applications where the presence of serial correlation is evident and can be accommodated using appropriate correlation structures (e.g. Morton and Henderson, 2008; Richards et al., 2013; Murphy et al., 2019; Yang and Moyer, 2020; Beck et al., 2022). To evaluate long-term trends, time is used as a smooth (or parametric) term. GAMs estimate the effect of time while accounting for other covariates, such as flow by modelling temporal patterns conditional on the other drivers included in the model. When water quality time series exhibit temporal autocorrelation, GAMs can be extended to Generalised Additive Mixed Models (GAMMs), which incorporate random effects and/or correlation structures (such as CAR1) to account for dependencies between sequential observations (Wood, 2017). While GAMs and GAMMs provide a robust approach to evaluate trends in water quality data, their outputs can be challenging to interpret due to the use of smoothing functions to capture non-linear relationships and random effects to account for autocorrelation. Therefore, their application to riverine water quality and associated flow data requires careful evaluation and clear communication to build confidence in trend analysis and to effectively quantify improvements in water quality that (may) result from improved catchment management.

In the tropical catchments draining to the Great Barrier Reef (GBR) World Heritage Area (GBRWHA) of northeastern Australia, land use changes over the past 150 years have resulted in elevated sediment and nutrient loads that threaten the World Heritage listed site (Brodie et al., 2012; Lewis et al., 2021; McCloskey et al. 2021a, 2021b). Two water quality constituents of particular concern to the health of the GBR are dissolved inorganic nitrogen (DIN) and suspended sediment (SS). Concentrations of these constituents are a key focus due to their ecological significance for organisms of concern. For example, increased riverine DIN concentrations have been associated with macroalgae overgrowth on coral reefs, greater vulnerability of corals to disease, enhanced crown of thorns starfish (COTS) outbreaks and thermal stress and light-induced bleaching (Fabricius, 2005; Wooldridge, 2009; De'ath and Fabricius, 2010; Haapkylä et al., 2011; Wiedenmann et al., 2013; Babcock et al., 2016; Brodie et al., 2017; Diaz-Pulido et al., 2024). Suspended sediments impact marine ecosystems such as seagrass meadows and inshore corals by reducing light penetration and smothering benthic organisms (Bartley et al., 2014; Bainbridge et al., 2018; Bozec et al., 2021; Collier et al., 2024). In response to these threats, the Australian and Queensland governments have implemented the Reef 2050 Water Quality Improvement Plan (hereafter referred to as the Reef 2050 Plan) which includes an investment of ~AU$5B from 2014 to 2030 (https://www.dcceew.gov.au/parks-heritage/great-barrier-reef/protecting/our-investments). The Reef 2050 Plan is particularly focused on land remediation (erosion control) and the encouragement and enforcement of improved land/farm management practices, including optimised fertiliser application and improved grazing land management (Kroon et al., 2016; Eberhard et al., 2021; Thorburn et al., 2024). Despite substantial investment, there has been limited evaluation of long-term trends in water quality concentrations across GBR catchments to assess the effectiveness of the Reef 2050 Plan.

The Queensland Government GBR Catchment Loads Monitoring Program (GBRCLMP) has extensively monitored 'end of catchment' sites within the GBR catchment area since the 2005/06 fiscal year (WQ&I, 2025a). These sites typically correspond to the most downstream monitoring flow gauges, capturing the quantity of nutrients and SS exported from the catchment to the GBRWHA. The program employs a rigorous quality assurance and quality control approach around sample collection, targeting both high and low flow periods as well as sample processing, analysis, and reporting (WQ&I, 2025b). Prior to this coordinated program, water quality monitoring in the GBR catchment area was conducted more sporadically and by different organisations and as such, the data were fragmented and dispersed across multiple individuals or institutions making access and integration challenging. These historical water quality data have recently been collated (James et al., 2025), and are available on the Tahbil water quality data portal (WQ&I, 2024). Across these two data resources, nine common sites were identified that each provide up to 37.5 years of non-continuous water quality monitoring to June 2024. The nine sites collectively cover 66% of the total GBR catchment area and their individual upstream catchments contain a mix of different land uses (including changing land uses over the monitoring period), catchment sizes, climatic regimes, hydrology, bioregions and different degrees of management intervention. A sophisticated statistical modelling approach is required to undertake trend analysis of these data to account for temporal autocorrelation and data gaps, as well as considering irregular and biased sampling (e.g. preferential sampling during high- or low-flow periods), thereby ensuring the robustness and statistical significance of detected trends.

This study applied GAMMs to long-term water quality data from nine monitoring sites across the GBR catchment area to examine linear and non-linear trends in DIN and SS concentrations. While assessments of potential marine impacts typically rely on total constituent loads - whose interpretation is confounded by the influences of variable runoff and hydrological conditions and warrants dedicated consideration in future work - this study prioritises an initial evaluation of temporal trends in constituent concentrations. Existing studies of riverine water quality trends have largely focussed on temperate environments, large river catchments, or are confined to single river systems (e.g. Broussard and Turner, 2009; Li et al., 2020; Chen et al., 2024; Botero-Acosta et al., 2025). While some recent work has expanded to multiple basins and broader hydroclimatic gradients (e.g. Costa et al., 2025; Guo et al., 2025), these studies do not explicitly focus on tropical settings nor fully capture the pronounced environmental heterogeneity that characterises them. Tropical river systems are among the most climatically and hydrologically variable in the world and are often subject to substantial land use change. Despite these characteristics, there remains a lack of multi-basin assessments that explicitly quantify linear and non-linear trends in water quality. To our knowledge, this is the first study to analyse linear and non-linear trends across multiple tropical river systems that collectively encompass substantial variability in catchment area, land use, climate and river regulation. The primary objective of this study is to use a robust and defensible methodology to quantify temporal trends in water quality accounting for non-linear patterns, episodic events, and step changes, answering the question: are river water quality concentrations in the GBR catchments changing over time? The results are interpreted in the context of climate variability, land use, and management practices to identify key drivers of change and locations exhibiting the most pronounced trends. These results provide empirical evidence supporting the evaluation of the Reef 2050 Plan's effectiveness and help guide future land management programs.

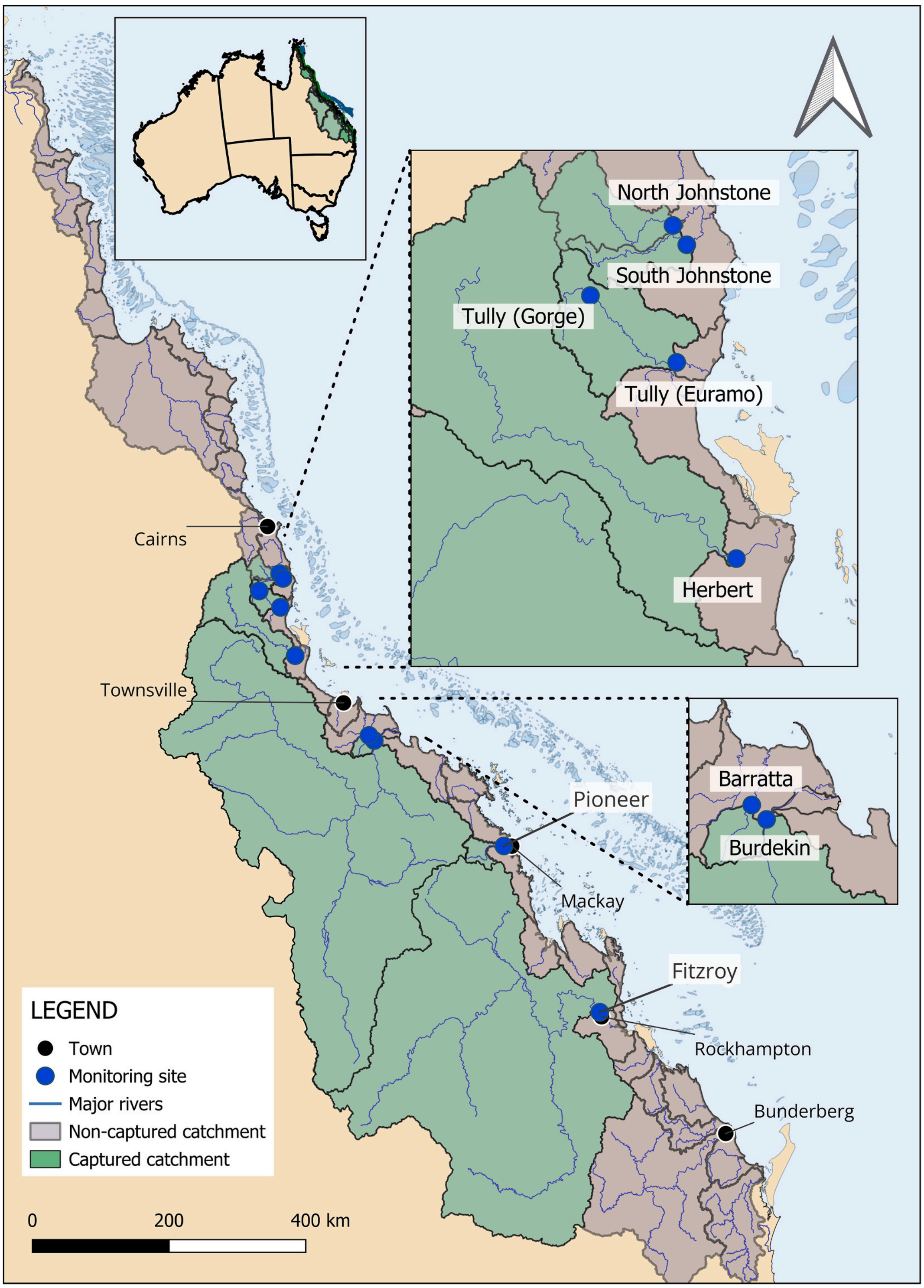

Fig. 1. Great Barrier Reef catchment area and the locations of the nine monitored sites (blue circles) and upstream catchment areas (shown in green).

Table 1
List of river monitoring sites from north to south including gauge details, annual discharge, rainfall, regulation, land use and major changes in the upstream basin area.

River	Gauge site name (Gauge No.)	Water quality monitoring location	Gauged drainage area (km^2)	Average annual rainfall (mm)[b]	Total average annual discharge (ML)[c]	Proportion total basin area (%) captured by gauge	Land use (%) upstream of monitoring location (2024)[d] - cropping, grazed, conservation (forest)	Major land use and changes in land use during the monitored period
North Johnstone	North Johnstone River at Tung Oil (112004A)	Goondi	925	2870	1,800,000	40%	4.2, 34, 51%	Rainforest, sugarcane, horticulture (bananas) and grazing. Decline in dairy cattle numbers over the past 30 years. Upstream catchment unregulated.
South Johnstone	South Johnstone River at Upstream Central Mill (112101B)	Upstream Central Mill	400	2970	790,000	17.3%	4.7, 8.8, 84%	Rainforest, sugarcane and horticulture (bananas). Minor land use change in upstream area over the past 30 years. Upstream catchment unregulated.
Tully	Tully River at Tully Gorge National Park (Daily modelled discharge[d])	Tully Gorge NP	480	2100	990,000	29%	0, 0.8, 90%	Rainforest. Koombooloomba Dam upstream (constructed 1960; 250 GL capacity) regulates 34% of upstream catchment ([a]CtoFR of 0.21). Logging ceased in 1988 but no major land use change upstream of the Gorge site over 30-year period. Increase in sugarcane upstream of Euramo in the early 1990s and bananas from the mid-1990s to the mid-2010s. Reduction in synthetic nitrogen fertiliser use from the mid-2010s (Lewis et al. In press). 11% of Euramo site upstream catchment regulated ([a]CtoFR of 0.07).
Tully	Tully River at Euramo (113006A)	Euramo	1450	2610	3,100,000	87%	13, 6.8, 75%	
Herbert	Herbert River at Ingham (116001E-F)	John Row Bridge	8581	1150	3,400,000	87%	3.6, 55, 33%	Mixed catchment with rainforest, grazing, historical mining and sugarcane. Reduction in synthetic nitrogen fertiliser use from the early 2000s (Lewis et al. In press). Upstream catchment unregulated.
Barratta	Barratta Creek at Northcote (119101A)	Northcote	753	790	190,000	19.7%	19, 76, 0.7%	Receives farm flood irrigation tailwater, sources from the Burdekin Falls Dam (BFD). Large land use change from the early 1990s with grazing converted to sugarcane due to new drainage infrastructure following BFD construction (1987). First sugarcane was planted c. 1990 with most farms in place by 2000.
Burdekin	Burdekin River at Clare (120006B)	Inkerman Bridge	129,900	620	9,000,000	99.5%	1.2, 88, 8.4%	Grazing dominated in upper catchment; sugarcane and irrigated horticulture (diverse fruit and vegetables) on lower floodplain. BFD (constructed 1987; 1860 GL capacity) regulates 89% of catchment ([a]CtoFR of 0.24).
Pioneer	Dumbleton Weir Tail Water (125016A)	Dumbleton Pump Station Headwater	1488	1460	870,000	89%	19, 33, 23%	Mixed catchment with upper catchment grazing/forest and lower catchment sugarcane-dominated agriculture. Multiple weirs including Dumbleton Rocks, Marian and Mirani and, the Teemburra Dam (latter constructed 1997; 148 GL capacity) which regulates 93% of the upstream catchment ([a]CtoFR of 0.18).
Fitzroy	Fitzroy River at The Gap (130005A)	Fitzroy River Water (1300040)	135,800	610	4,900,000	95%	4.7, 78, 7%	Grazing dominated with cropping (grain/cotton) and major coal mining in the upstream catchment. A series of dams and weirs upstream including Fairbairn Dam (constructed 1972; 1301 GL capacity) regulates 100% of upstream catchment ([a]CtoFR of 0.27).

Daily modelled discharge from the Source Catchments model (McCloskey et al., 2021a) due to gauge 113015A only opening partway through time series.
[a] CtoFR is the capacity to flow ratio, which is the capacity of the dam (ML) divided by mean annual flow and is a surrogate for residence time.
[b] SILO data 1990-2024 (SILO; https://www.longpaddock.qld.gov.au/silo/index.html).
[c] WMIP: time period: 1990/1991 to 2019/2020.
[d] Water Quality and Investigation (2025a) based on QLUMP (2021).

2. Methods

2.1. Study sites

The GBR catchment area straddles the wet and dry tropics of northeastern Australia, occupying approximately 424,000 square kilometres (Fig. 1). The catchment consists of thirty-five river basins which drain into the adjacent Great Barrier Reef lagoon and hosts a considerable diversity of climate/rainfall, geology, vegetation, land use and levels of river regulation (Furnas, 2003; Lewis et al., 2021; Mariotti et al., 2021). The hydrology of the region is characterised by highly variable inter-annual and intra-annual discharge along a pronounced north-south latitudinal gradient (Puignou Lopez et al., 2025). Hence, detecting trends amongst this "noise" is challenging and generally, the greater the variance, the longer the temporal dataset needs to be to detect any underlying trend (Darnell et al., 2012; Bainbridge et al., 2014). This is particularly so for dry tropical river systems where higher inter- and intra-annual flow variability is manifested in seasonal peaks and troughs and influenced by the El Niño-Southern Oscillation and the passage of tropical cyclones (Rustomji et al., 2009; Puignou Lopez et al., 2025).

Following a review of the available water quality data across the GBRCLMP and the complementary historical datasets (see James et al., 2025), nine sites were selected within seven river basins for the trend analysis (Fig. 1; Table 1). Sites were selected that had monitoring records that span the longest time periods, while also representing a latitudinal gradient that captures the distinct inter- and intra-annual flow variability of the GBR catchment area (Table 2). Additional selection factors included sites that had limited land use changes in their upstream catchment area as a reference site (i.e. Tully Gorge) to compare with sites that had considerable land use changes over the monitoring period (i.e. Barratta). The selected sites also captured a wide range of wet and dry tropical climates, upstream catchment areas and different management interventions (targeting DIN and/or sediment reductions) in the upstream reaches.

Four out of the nine sites (North Johnstone, South Johnstone, Tully Gorge and Tully at Euramo) are in the Wet Tropics bioregion and are characterised by mean annual rainfalls > 2000 mm, with a consistent wet season between December and April (Table 1). These sites are generally represented by grazing, dairy or rainforest in their upper catchments and sugarcane cropping lands on the lowlands adjacent to the coast (Table 1). A fifth site (Herbert), lies on the boundary between the Wet Tropics and the seasonally Dry Tropics to the south, and has a correspondingly lower mean annual rainfall of approximately 1150 mm. The upstream land use within each of the sites range from 90% rainforest and no fertilised cropping at the Tully Gorge site, to $< 50\ km^2$ of fertilised cropping lands (North Johnstone and South Johnstone sites) to $>150\ km^2$ of fertilised cropping (Tully at Euramo and Herbert). Both the Herbert and North Johnstone sites have considerable tableland or escarpment terrain in their upper catchments that host both grazing and cropping, while the Tully and South Johnstone sites largely contain steep rainforest in their upper catchment reaches. The Wet Tropics bioregion is a priority area for DIN management due to elevated fertiliser losses from the cropping lands, while the management of sediment erosion is a priority in the upper catchments of the Johnstone and Herbert basins (McCloskey et al., 2021a, 2021b; Waterhouse et al., 2025).

The Burdekin and Fitzroy are the two largest basins collectively representing 64% of the GBR catchment area and, in combination, contribute 55% of the total annual sediment load to the GBR (McCloskey et al., 2021a). These large ($>$130,000 km^2) dry tropical basins have lower and more sporadic rainfall seasons (average annual rainfall of ~600 mm) and are dominated by rangeland cattle grazing (Table 1). Sugarcane and horticulture crops are located on the lower floodplain area of the Burdekin River delta. The Burdekin site is highly regulated by the Burdekin Falls Dam which captures 89% of the upstream catchment area, although its capacity to inflow ratio (0.24; which is the capacity of the dam in megalitres, divided by mean annual flow) indicates that the water residence times in the dam are relatively short; on average the dam traps 66% of the sediment that is transported from the catchments above the reservoir (Lewis et al., 2013, Table 1). While there are several dams and weirs in the upstream reaches of the Fitzroy site, the collective small volumes of these reservoirs mean they also have relatively short residence times (capacity to inflow ratio of ~0.27; Waterhouse et al., 2024) limiting their ability to capture sediments and process and breakdown nutrients.

The Barratta and Pioneer sites represent smaller coastal basins with high proportions (>18%) of sugarcane in their upper catchments and have highly variable rainfall with annual averages of 790 and 1460 mm, respectively (Table 1). Other dominant land uses in these catchments include forest (Pioneer) and coastal grazing (Barratta) (Table 1). Importantly, water quality monitoring at the Barratta site captured the rapid expansion of irrigated sugarcane cropping in the catchment which commenced in the early 1990s following the construction of the Burdekin Falls Dam in 1987. Prior to the introduction of sugarcane, land use in Barratta Creek was dominated by cattle grazing. The Pioneer site is also somewhat regulated by several dams and weirs along the river, although the water residence times are again relatively short (capacity to inflow ~0.18) (Waterhouse et al., 2024).

2.2. Water quality and discharge data

Eight of the selected sites for the trends analysis have water quality records spanning more than thirty years with one additional site in the Pioneer River catchment having a record spanning 18 years (Table 2 and Fig. 1). These data were collected under the Queensland Government's GBRCLMP (2006-current), augmented with historical data collated from several programs outlined in James et al. (2025). Early water quality records compiled by James et al. (2025) are fragmented with notable data collection from multiple sites through the Australian Institute of Marine Science (AIMS) catchment sampling program (1986-2000) (Furnas, 2003; AIMS, 2007) and the Queensland Environmental Protection Agency ambient water quality monitoring program (1992-2001) (Cox et al., 2005). Targeted programs for individual river catchment sites such as the Johnstone River (Hunter et al., 2001; Hunter and Walton, 2008), Tully River (Laxton and Gittins, 2004; Bainbridge et al., 2009; WTMIP, 2023), Herbert River (Bramley and Roth, 2002), Barratta Creek (Bainbridge et al., 2008 and references therein; Novic et al., 2018), Burdekin River (Bainbridge et al., 2006), Pioneer River (Rohde et al., 2008) and Fitzroy River (Packett et al., 2009) contributed further data spanning multiple years. All of these data are available on the public Queensland Government Tahbil - water quality data portal (WQ&I, 2024).

Additional SS data (for all sites except Tully River Gorge) were included from various state government programs (IAS, MRHI, SMP, SWAN) housed on the public Queensland Government Water Monitoring Information Portal (WMIP) (Department of Local Government, Water and Volunteers, 2025). Owing to the strong bias of the sampling regimes of most of these programs to low flow (ambient) conditions, these data were only used in combination with other datasets compiled (as per last paragraph) for our trend analysis in the years that contained a range of flow variability. Furthermore, with the exception of the Barratta Creek site, the DIN data collected under these programs were excluded due to insufficient documentation and metadata on sample collection, storage and laboratory procedures; consequently, we lack confidence in the quality of the data for these volatile nutrient species. This follows data exclusion criteria outlined in James et al. (2025). The Barratta Creek DIN data housed on the WMIP were collected under the Barratta Wetlands Study (BWS) with details on sample collection and analysis well documented in Congdon and Lukacs (1995). The Barratta Creek DIN time series was further supplemented with data transcribed from a Bureau of Sugar Experiment Stations report (Ham, 2006) for this study.

Laboratory analysis was performed using standard wet chemical approaches for oxidised nitrogen and ammonium as detailed in WQ&I, 2025b and James et al. (2025). DIN was determined as the sum of oxidised nitrogen (NO_2+NO_3) and ammonium (NH_4). In cases where only nitrate and ammonium were available (no nitrite), the samples were still included in this analysis as nitrite concentrations are typically very low in freshwaters because it rapidly oxidises to nitrate under aerobic conditions. Below detection values were halved prior to summing. SS is measured as total suspended solids, determined gravimetrically using the APHA (2005) standard approach. We note the laboratories utilised by the different water quality monitoring projects have applied filter pore sizes ranging from 0.7 to 1.5 μm for SS (James et al., 2025). All the data included in this analysis have been subject to QA/QC processes (for further details refer to WQ&I, 2025b and for historical data refer to the project specific metadata statements in James et al. (2025)).

Daily measured stream discharge data (m^3/s) were downloaded from the WMIP (WMIP:water-monitoring.information.qld.gov.au; DLGW&V, 2025) to match the date range for the water quality sample data collected from each site (Table 1). Except for two sites, the stream flow data represented continuous daily measurements over the entire period. A three-month discharge data gap (11/02/2004-23/04/2004) for the South Johnstone at Upstream Central Mill gauge site (112101B) was augmented with estimated flow for this period using a GAM-fitted relationship with the nearest Bureau of Meteorology river height station 14 km upstream (BOM station 032,161). River flow gauging at Tully Gorge site only commenced in November 2009. Therefore, at this site we used modelled daily flow from the Paddock to Reef Program Source Catchments hydrology model (McCloskey et al., 2021a).

2.2.1. Data quality assessment and inclusion criteria

A key issue with water quality trend analysis is ensuring sample representativeness and avoiding biased estimates of constituent responses (Thompson et al., 2021). Concentrations can change rapidly in river flows, and many constituents are predominantly transported during 'first flush', high flow or event conditions (Hunter and Walton, 2008; Novic et al., 2018). Sampling biased to low flow conditions are therefore likely to miss periods of elevated concentrations and may present very different results to that from samples collected during high flow conditions. This is important in the context of the GBR, where flows are highly variable, and constituent concentrations can vary by orders of magnitude over a flow event and over successive events (Hunter and Walton, 2008; Kuhnert et al., 2012; Novic et al., 2018).

Several water quality studies have applied trend analysis to different

Table 2
Summary of collated water quality datasets and DIN and SS concentration statistics (raw data). Sample numbers (*n*) represent raw data, with the number of DIN or SS values averaged to a daily time step for GAMM analysis presented in brackets below. Water quality data project sources and citations are also included, along with historical project codes as detailed in the James et al. (2025) data compilation. In cases where monitoring locations shifted slightly over time and between different programs, data from these locations were combined, and the full list of sites and their corresponding locations is provided in Supplementary Table 1.

Water quality monitoring site name (short name)	DIN collated dataset				SS collated dataset				Water quality data sources (original projects and key references)
	n	Date range	concentration (mg/L)		*n*	Date range	concentration (mg/L)		
			Median	Range			Median	Range	
North Johnstone River at Goondi	*1120 (793)*	28/07/1988-30/06/2024	0.124	0.002-0.595	*1160 (816)*	20/12/1989-30/06/2024	12	0.4-504	GBRCLMP (WQ&I, 2024); AIMSGBR (AIMS, 2007); NHTJR (Hunter et al., 2001; Hunter and Walton, 2008). *SS data only:* Water Monitoring Information Portal (WMIP) - various government projects (DLGW&V, 2025)
South Johnstone River at Upstream Central Mill	*1621 (1176)*	14/12/1986-30/06/2024	0.132	0.004-1.063	*1354 (960)*	01/02/1987-30/06/2024	15	0.5-7,638	GBRCLMP (WQ&I, 2024); AIMSGBR (AIMS, 2007); NHTJR (Hunter et al., 2001; Hunter and Walton, 2008). *SS data only:* WMIP - various government projects (DLGW&V, 2025)
Tully River at Tully Gorge NP (Tully Gorge)	*1143 (725)*	01/09/1987-30/06/2023	0.068	0.006-1.046	*1063 (597)*	01/01/1991-30/06/2023	12	0.5-1,250	GBRCLMP (WQ&I, 2024); AIMSGBR (AIMS, 2007); EPAGBR (Cox et al., 2005); LAXTGBR (Laxton and Gittins, 2004); TWQIP (Bainbridge et al., 2009); WTMIP (Wet Tropics Major Integrated Project, 2023)
Tully River at Euramo	*2795 (1536)*	18/09/1987-30/06/2024	0.194	0.006-2.126	*2640 (1294)*	19/02/1996-30/06/2024	25	0.5-447	GBRCLMP (WQ&I, 2024); AIMSGBR (AIMS, 2007); EPAGBR (Cox et al., 2005); LAXTGBR (Laxton and Gittins, 2004 – Note DIN data were only included for 1998-2000 where it augmented existing datasets); TWQIP (Bainbridge et al., 2009). *SS data only:* WMIP - various government projects (DLGW&V, 2025)
Herbert River at John Row Bridge	*1455 (1197)*	28/07/1988-30/06/2024	0.124	0.005-2.147	*1439 (1157)*	01/09/1990-30/06/2024	26	0.5-904	GBRCLMP (WQ&I, 2024); AIMSGBR (AIMS, 2007); CSIROHR (Bramley and Muller, 1999; Bramley and Roth, 2002). *SS data only:* WMIP - various government projects (DLGW&V, 2025)
Barratta Creek at Northcote	*2050 (1097)*	13/04/1988-30/06/2024	0.220	0.002-18.06	*1962 (1140)*	01/03/1989-30/06/2024	44	0.5-11,000	GBRCLMP (WQ&I, 2024); BWS (Congdon and Lukacs, 1995); LBWQIP (Bainbridge et al., 2008). *DIN data only:* Ham, 2006; RRRDBAR (Novic et al., 2018). *SS data only:* WMIP - various government projects (DLGW&V, 2025)
Burdekin River at Inkerman Bridge	*1597 (1362)*	17/12/1987-30/06/2024	0.115	0.002-0.780	*1284 (1092)*	06/01/1996-30/06/2024	144	0.5-3,559	GBRCLMP (WQ&I, 2024); AIMSGBR (AIMS, 2007); BCWQ (Bainbridge et al., 2006); EPAGBR (Cox et al., 2005). *SS data only:* WMIP - various government projects (DLGW&V, 2025)
Pioneer River at Dumbleton	*1365 (725)*	23/01/2006-30/06/2024	0.181	0.002-3.557	*1396 (743)*	23/01/2006-30/06/2024	21	0.5-1,680	GBRCLMP (WQ&I, 2024); MWHW08 (Rohde et al., 2008). *SS data only:* WMIP - various government projects (DLGW&V, 2025)
Fitzroy River at Fitzroy River Water (Rockhampton)	*955 (748)*	14/11/1992-30/06/2024	0.206	0.002-0.749	*1023 (771)*	18/08/1992-30/06/2024	161	0.5-2,325	GBRCLMP (WQ&I, 2024); AIMSGBR (AIMS, 2007); NAPFR (Packett et al., 2009). *SS data only:* WMIP - various government projects (DLGW&V, 2025)

Table 3
Summary of the hydrological explanatory terms (discounted flow terms previously described by Kuhnert et al. (2012)).

Term description	Time period represented	Term name(s)
Trend term	Entire time series of data, annual time step	trendY
Flow term (log)	Entire time series of data	pQ
Seasonal term	Within a year	DoY
Rising and falling limb terms	Event based on 90th percentile of flow	limb
Hydrological dynamics (discounted flow) terms		
• Short term	2 days	MA2days
• Medium term	1 week	MAweek
• Long term	1 month	MAmonth
	6 months	MA6months
	12 months	MA12months

flow conditions after separating low (or base) flow from high (or event) flows to isolate the different pathways and time scales through which constituents are preferentially transported. These studies have used a variety of visual/graphical, numerical/digital, isotopic and/or hydrological approaches to separate the different flow components (e.g. Zhang et al., 2021; Chen et al., 2024). Although baseflow separation was initially explored in this study, there were considerable challenges implementing a consistent approach across basins with highly variable intra- and inter-annual hydrology. The hydrological variability, compounded by the paucity of concentration data during certain historical intervals, complicated this analysis. Hence, it was determined that a single overall trend analysis was most appropriate to maximise the available information and improve our capacity to generalise model results.

It is important to acknowledge that biases in water quality data can also arise through other factors. For example, changing from manual to automated sampling can affect both sampling frequency and sampling location; autosamplers generally collect samples at the edge or mid-channel close to the stream bed, whereas manual sampling often occurs at the edge of the waterway and near the surface of the water column. Early data from some sites included SS data collected using a smaller pore sized filter paper than standard for freshwaters, thus increasing the proportion of finer solids captured. These finer solids are generally a small proportion of the total sediment concentrations and are unlikely to influence the results (Bainbridge et al., 2016).

In a preliminary step, this study explored potential sampling biases arising from the timing of sample collection in relation to median flow conditions for all available data (Supp. Fig. 1). Bias was calculated by dividing the flow recorded at the time the concentration samples were collected (DIN or SS) by the median flow recorded for that fiscal year (Kuhnert et al., 2012). Bias was explored in both the GBRCLMP and historical datasets (Supp. Figs. 2 and 3). The plots demonstrate sampling bias to elevated flow periods particularly for the dry tropical rivers with higher inter-annual flow variability (Supp. Figs. 2 and 3). It also demonstrates a sample bias in the collection of the GBRCLMP datasets to higher flow conditions relative to the historical datasets, although this varies across the different sites. Such bias is not unexpected given the intent of the GBRCLMP program to determine river pollutant loads (Thomson et al., 2012).

The identification and selective removal of water quality data which exhibit a clear bias is a fundamental requirement prior to trend analysis. However, achieving this balance requires some care, given the paucity of historical data to ensure any data refinement does not compromise objectivity or the ability to generalise the approach to other sites. There was a need to carefully balance the exclusion of data, while ensuring valuable information was retained. The inclusion of water quality sample data for trend analysis in this study was determined using a set of key considerations aligned with several principles for best practice monitoring design. These include ensuring that:

1. There is confidence in data quality supported by comprehensive metadata documenting sample collection, storage and laboratory procedures;
2. Samples are representative of the flow conditions under which constituents are delivered. In this study, data exploration was undertaken through visual inspection (Supp. Figs. 1–3) and review of model diagnostics. As part of this process outliers were examined but ultimately, we decided to retain these values in all our datasets. The GBR catchments have high variability in concentrations across varying flow conditions and there was not sufficient justification to remove these extreme values;
3. There is adequate data coverage across the study period, including at the temporal boundaries, to support stable estimation of the GAMM smoothing terms, minimise edge effects, and ensure that estimated trends reflect genuine long-term change rather than artefacts arising from model flexibility (Wood, 2017; Simpson, 2018). For some sites we truncated the data series by one to two years to ensure we had representative data coverage.

All the datasets in this study displayed some bias. In some cases, bias in datasets measuring constituent concentrations may be intentional, reflecting targeted monitoring objectives. However, this represents a key challenge in curating datasets collected through different monitoring programs over varying timeframes. In most cases, bringing in multiple different datasets enriched the historical record. Earlier programs were focused on either event flows to calculate event loads (as this is important for understanding the impacts upon the GBR) or on low flows to assess stream water quality during ambient flow conditions (which is particularly important for their freshwater ecology). Hence, in combination, the datasets represented both event and low flows and provided a reasonable representation of the concentrations across the range of flow conditions.

2.3. Statistical analysis

The first stage of the statistical analysis involved the curation of the concentration and flow data, where the irregular concentration data were matched to the flow data, captured at regular intervals (e.g. daily). Note, flow data from the gauge are generally regular, and in instances where flow was missing, an appropriate interpolation approach using a LOESS smoother was used to infill. Where multiple concentration data were available within a flow time step, these data were averaged. Explanatory terms were then created from the flow data to represent the dynamic hydrological characteristics typical of the GBR river basins (Table 3). The specific terms are described by Kuhnert et al. (2012) and are available in the Loads Regression Estimator (LRE), an R package for modelling concentration and flow relationships and calculating loads (https://pkuhnert.github.io/LRE/). Terms considered for this analysis included a seasonal (periodic) component, a flow term, rising-falling limb terms that capture concentration changes that typically occur on the rise or the fall of the peak event and, hydrological terms that mimic short term, intermediate-term and long-term memory of the system. These latter terms are computed as exponential moving averages using the TTR package in R (Ulrich, 2026), and are based on 2 days, 1 week, 1 month, 6 month and 12 month periods (Table 3). They represent an exponential moving average of flow history intended to capture the influence of flow over realistic lagged time periods. An exponential moving average gives more weight to the most recent flows and less weight to those in the past.

A GAMM was developed for each site and each water quality parameter (DIN or SS) to model the relationships in water quality over time. The models were fitted using the 'gamm' function in the 'mgcv' package (Wood, 2017), which also depended on the 'nlme' package (Pinheiro and Bates, 2000; Pinheiro et al., 2025) to fit random effects. Through the additive nature of the GAMM, each term's contribution in the model can be viewed conditionally on all other terms in the model.

Table 4
Non-linear (episodic) and linear trend results for dissolved inorganic nitrogen (DIN) concentrations at nine sites within the GBR catchment area. Where linear trends are significant, the annual decrease or increase in concentration is provided.

Site	Date range	Trend time period (years)	Non-linear trend (i.e. episodic)		Linear trend across the time series (Box-Cox transformed)				
			Non-linear trends	GAMM model – adjusted R^2	Significant concentration trend over time	Median (mg/L) – first 10 years (historical)	Instantaneous rate of change (95% CI)	*P*-value	GAMM model – adjusted R^2
North Johnstone River at Goondi	Jul 1988 - Jun 2024	36	✓	0.65	Decreasing trend	0.097	0.61% (0.27 to 0.94) decline per year	<0.001	0.64
South Johnstone River at Upstream Central Mill	Dec 1986 - Jun 2024	37.5	✓	0.67	Decreasing trend	0.109	1.3% (1.0 to 1.6) decline per year	<0.001	0.63
Tully River at Tully Gorge National Park	Sep 1987 - Jun 2023	36	✓	0.49	Decreasing trend	0.053	1.0% (0.37 to 1.6) decline per year	0.002	0.18
Tully River at Euramo	Sep 1987 - Jun 2024	37	✓	0.45	Decreasing trend	0.158	0.43% (0.12 to 0.74) decline per year	0.007	0.43
Herbert River at John Row Bridge	Jul 1988 – Jun 2024	36	✓	0.26	Decreasing trend	0.130	1.2% (0.72 to 1.7) decline per year	<0.001	0.25
Barratta Creek at Northcote	Apr 1988 - Jun 2024	36	✓	0.44	Increasing trend	0.038	4.7% (3.9 to 5.5) increase per year	<0.001	0.40
Burdekin River at Inkerman	Dec 1987 - Jun 2024	36.5	✓	0.65	Decreasing trend	0.182	2.7% (2.2 to 3.2) decline per year	<0.001	0.45
Pioneer River at Dumbleton	Jan 2006 - Jun 2024	18.5	✓	0.51	Decreasing trend	0.166	3.0% (1.0 to 5.0) decline per year	0.003	0.43
Fitzroy River at Fitzroy River Water	Nov 1992 - Jun 2024	31.5	✓	0.55	*Not significant*	-	-	0.97	0.50

Table 5
Non-linear (episodic) and linear trend results for suspended sediment (SS) concentrations at nine sites within the GBR catchment area. Where linear trends are significant, the annual decrease or increase in concentration is provided.

Site	Date range	Trend time period (years)	Non-linear trend (i.e. episodic trend)		Linear trend across full time series (Box-Cox transformed)				
			Non-linear trends	GAMM model –adjusted R^2	Significant concentration trend over time	Median (mg/L) – first 10 years (historical)	Instantaneous rate of change (95% CI)	*P*-value	GAMM model – adjusted R^2
North Johnstone River at Goondi	Dec 1989 - Jun 2024	34.5	✓	0.74	Decreasing trend	7	1.6% (1.0 to 2.1) decline per year	<0.001	0.74
South Johnstone River at Upstream Central Mill	Feb 1987 - Jun 2024	37	✓	0.61	*Not significant*	-	-	0.78	0.60
Tully River at Tully Gorge National Park	Jan 1991 - Jun 2023	32.5	✓	0.56	Decreasing trend	12	6.3% (5.1 to 7.5) decline per year	<0.001	0.51
Tully River at Euramo	Feb 1996 - Jun 2024	28.5	✓	0.67	Increasing trend	9	0.71% (0.02 to 1.4) increase per year	0.045	0.65
Herbert River at John Row Bridge	Sep 1990 - Jun 2024	33.5	✓	0.73	Decreasing trend	51	2.9% (2.2 to 3.5) decline per year	<0.001	0.72
Barratta Ck at Northcote	Mar 1989- Jun 2024	35	✓	0.64	Decreasing trend	21	1.4% (0.86 to 1.9) decline per year	<0.001	0.59
Burdekin River at Inkerman	Jan 1996 - Jun 2024	28.5	✓	0.79	Decreasing trend	238	4.5% (3.8 to 5.2) decline per year	<0.001	0.76
Pioneer River at Dumbleton	Jan 2006 - Jun 2024	18.5	✓	0.71	Decreasing trend	49	5.9% (4.3 to 7.4) decline per year	<0.001	0.70
Fitzroy River at Fitzroy River Water	Aug 1992 - Jun 2024	31.5	✓	0.73	Decreasing trend	225	1.3% (0.23 to 2.4) decline per year	0.018	0.63

This study used a cyclic cubic spline smoothing function for the seasonal term, and a cubic spline smoothing function for other terms. For the seasonal term, 20 spline knots were equally positioned across the 365 or 366 days of the year to ensure the term was cyclical. The trend was fitted either as a linear term or as a non-linear smoothing spline to allow for examination of both linear and non-linear (or episodic) temporal patterns. The response terms (DIN or SS) were transformed using a Box-Cox transformation (Box and Cox, 1964) to stabilise variance and normalise the distribution of the responses, noting that as the transformation parameter, lambda (λ), approaches zero, the resulting function converges to the natural logarithm.

Due to the high frequency of concentration measurements during high flow conditions, temporal autocorrelation was anticipated and detected in all models. Autocorrelation reduces the effective sample size by creating dependencies between observations, yet standard methods that assume independence treat the full nominal sample size as if it contained independent information. This mismatch produces underestimated standard errors and overly narrow confidence intervals, which in turn can generate overconfident predictions and spurious findings of significance (Heffernan et al., 2007; Leigh et al., 2019; Yang and Moyer, 2020). When residual autocorrelation persists, parameter estimates for flow terms may be biased, as temporal dependence is mistakenly attributed to predictor effects (Cochrane and Orcutt, 1949). To account for the autocorrelation, models were fitted with a random effect that incorporated a continuous autoregressive first-order (corCAR1) structure. This approach explicitly models the dependency between consecutive observations within an irregular time series, while maintaining the flexible non-linear relationships captured by the smoothing terms.

After fitting the full models including all candidate terms (Table 3), model simplification was undertaken using an Akaike's Information Criterion (AIC)-based selection approach. Beginning with the full model, each predictor was sequentially removed and the model refitted. Core terms representing flow and season (pQ and DoY) and, trendY (smoothed or linear depending on the specific model) were retained regardless of their significance, as these represent fundamental drivers of flow constituents such as nutrients and sediments in the GBR river catchments and provide the structural basis of the model. Model performance was evaluated by comparing AIC values between the full and reduced models. At each step, the term whose removal resulted in the greatest improvement (i.e. the largest decrease in AIC) was excluded. This procedure was repeated iteratively, with terms removed where

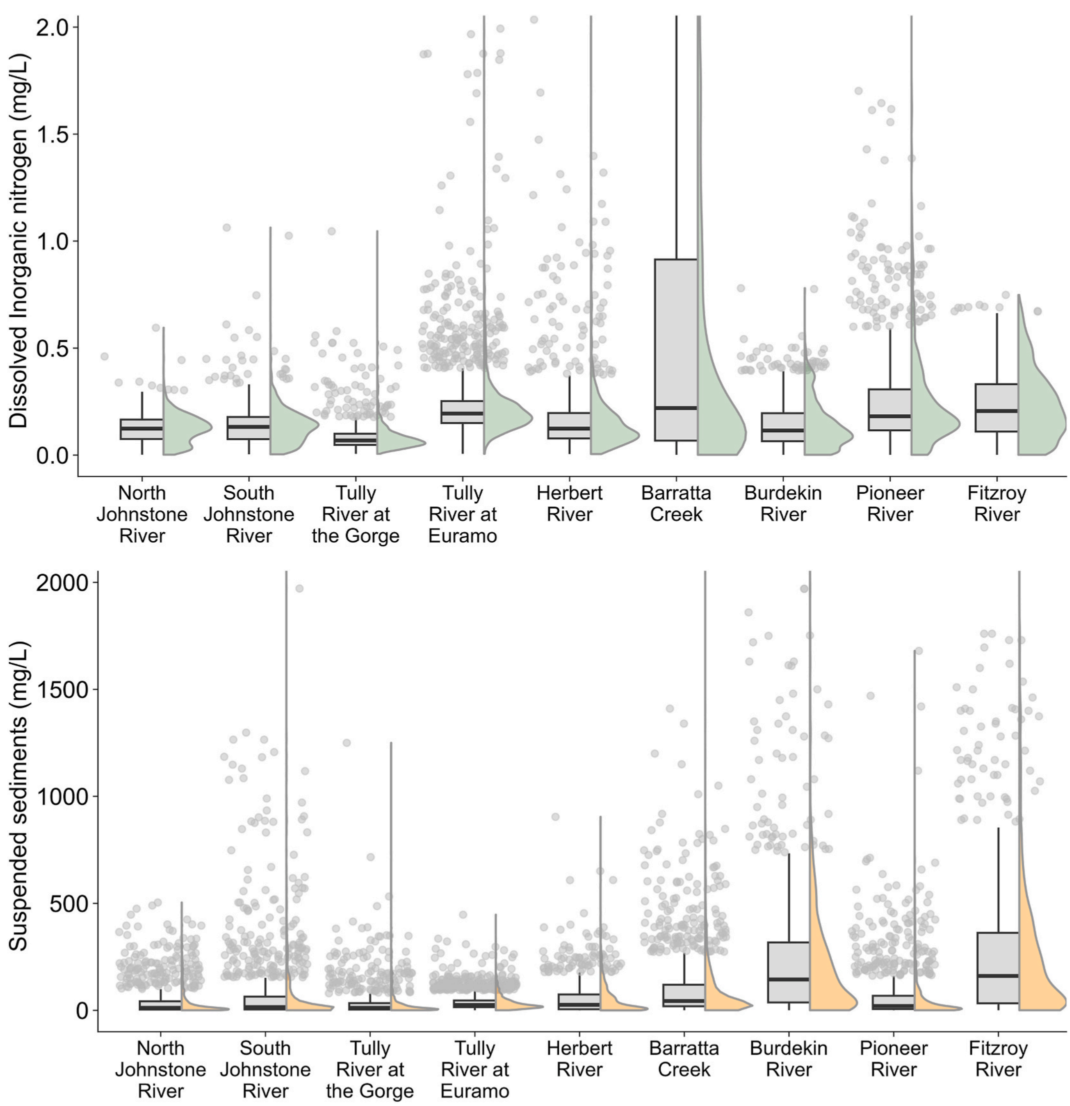

Fig. 2. DIN and SS concentration box and whisker plots based on the raw water quality data observations across the nine sampling sites (black lines represent medians, the box extents are the interquartile ranges, and the points represent observations above the 95th percentile, highlighting the more extreme observations within each site's distribution. Violin plots (green and orange) are also included to represent concentration distribution of each data series. DIN and SS observations have been truncated for visualisation (>2 mg/L and >2000 mg/L, respectively); refer to Table 2 for full concentration ranges.

their exclusion resulted in an improvement of at least two AIC units consistent with the threshold for 'substantial support' described by Burnham and Anderson (2002). This iterative process continued until no further reductions meeting this criterion were achieved. The final model was selected as the most parsimonious model with the lowest AIC.

Following selection of the final models, we explored both linear and non-linear temporal trends. Significant non-linear (episodic) trends were explored visually using a computational approach based on derivatives and first differencing that accounts for model uncertainty to highlight the significant periods of change along the response curve (Simpson, 2018). Significant periods of change were identified where the 95% confidence interval for the estimated first derivative did not include zero. The interpretation of the linear trend term where a Box–Cox transformation has been applied can be complex when $\lambda \neq 0$. This is because the back-transformed relationship between time and the response is non-linear, and the effect of the trend term should not be

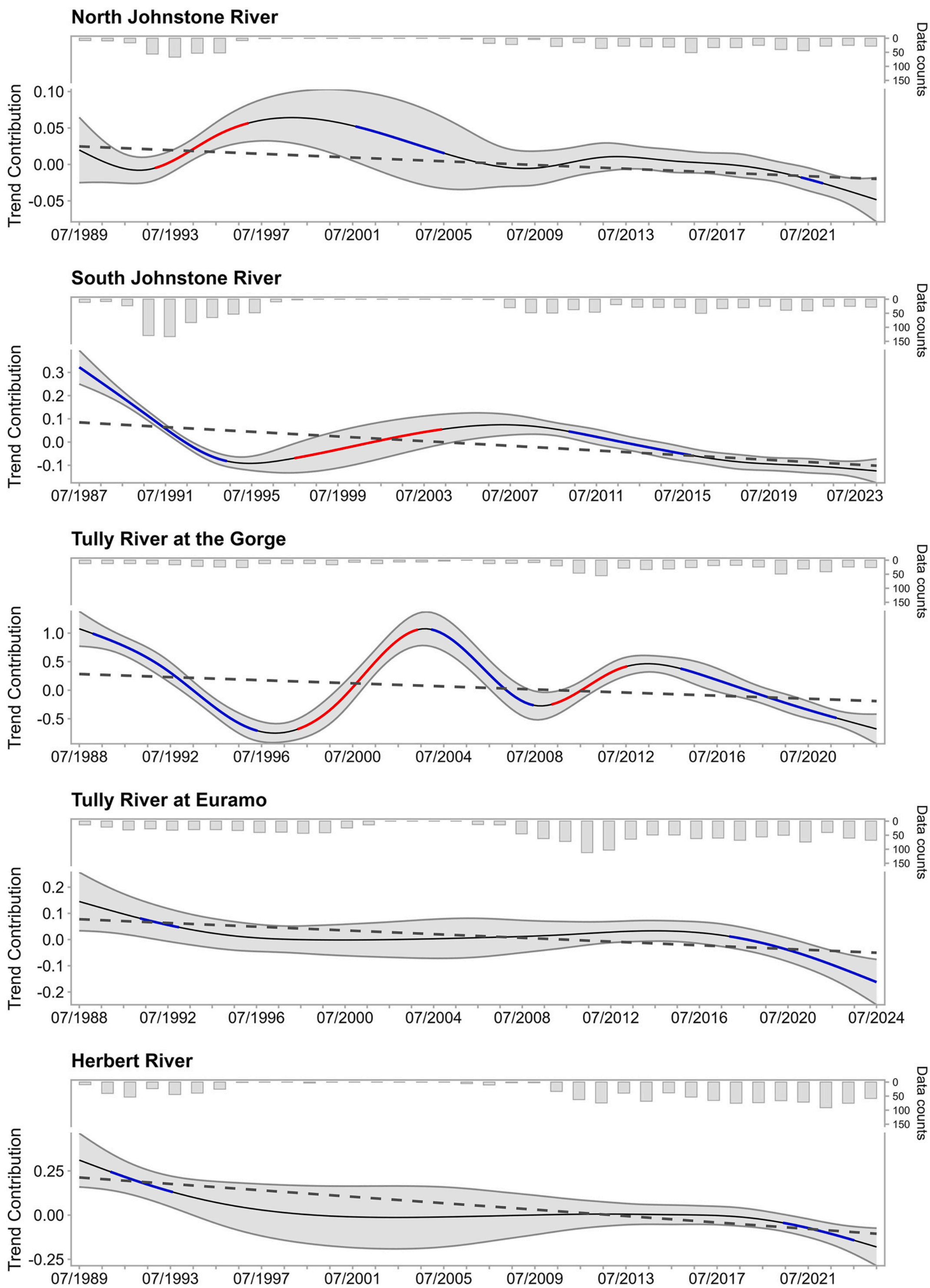

Fig. 3. Non-linear and linear trends in dissolved inorganic nitrogen across the nine sites. Significant (episodic) periods are represented as red lines (increasing) and blue lines (decreasing). Significant linear trends are also plotted as dashed line. The hanging bar chart provides concentration data sample numbers for each fiscal year.

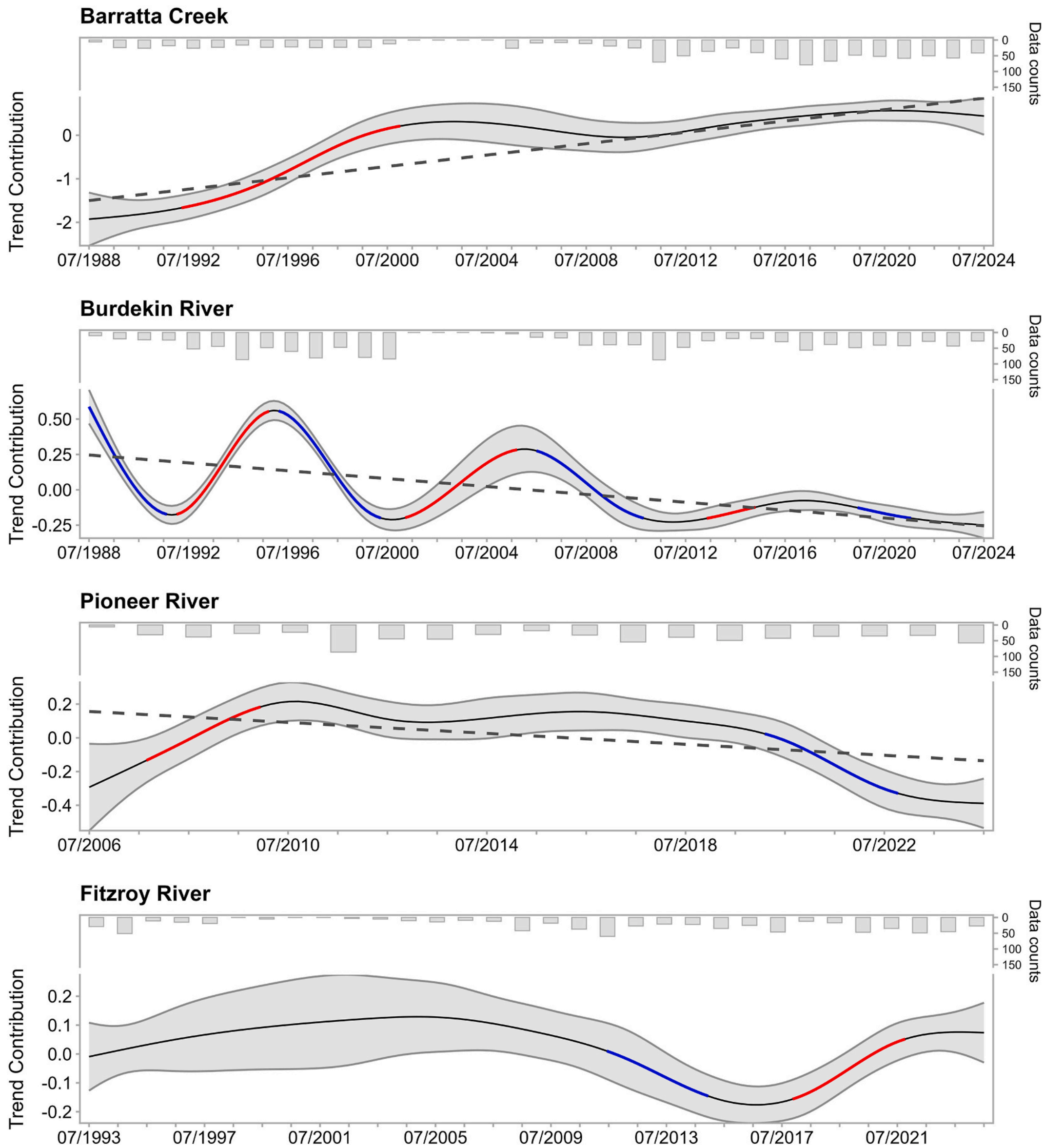

Fig. 3. (*continued*).

interpreted as a constant annual change on the original scale. Instead, the rate of change varies with concentration. We estimated rates of change from the first derivative of the back-transformed Box–Cox function, providing concentration-dependent instantaneous estimates of the rate of increase or decrease per year. The median concentration of the first ten years of each record was used as the reference value for estimating these rates. The first ten years were considered representative of average conditions in these highly episodic tropical environments. Both the non-linear and linear trend results for the sampling period are presented when significant.

Additional R packages were used for specific tasks: 'MASS' (Venables and Ripley, 2002) for Box-Cox data transformation, 'gratia' (Simpson, 2024) for exploring and visualising smoothing terms, 'itsadug' (van Rij et al., 2022) for generating autocorrelation function (ACF) plots, 'ggplot2' Wickham, 2016 for creating additional figures, and 'patchwork' Pedersen, 2020 for arranging figures.

3. Results

3.1. GAMM performance across the nine sites

Model performance was assessed using adjusted R^2, alongside inspection of diagnostic plots (including residual, QQ, and autocorrelation plots) to evaluate model assumptions and residual structure (Tables 4 and 5; see also specific model results provided in Supp. Tables 2–19). To address non-normality and heteroscedasticity, response variables were transformed using a Box–Cox transformation, with the optimal lambda (λ) parameter estimated for each site and metric combination. The range of λ values (−0.101 to 0.667 for DIN; −0.141 to 0.182 for SS) shows moderate departures from normality, with transformations ranging from near-log ($\lambda \approx 0$) to modest stabilisation of variability (Supp. Tables 2–19). Negative λ values indicate particularly heavy-tailed or strongly right-skewed data, requiring transformations stronger than the log to achieve approximate normality. DIN λ values showed a wider range across sites, while SS values were more tightly clustered around zero, suggesting that a log transformation works consistently well for SS across sites, reflecting moderate and consistent right-skew. Because we used a Box–Cox transformation on the response, linear trends on the transformed scale do not translate to a constant linear trend on the original scale. However, for most sites, the combination of small trend coefficients and λ values close to zero means the fitted trend is approximately linear on the original scale over the study period.

The models explain quite different proportions of the variation in the responses (from 0.26 to 0.79) noting that adjusted R^2 have a more nuanced interpretation for GAMMs compared with linear models. Whilst a model with a lower R^2 does not necessarily mean a poorer model (high R^2 can imply over fitting), it does suggest variability in the response that is not being captured by the explanatory variables included in the model. The adjusted R^2 values for the DIN non-linear models ranged between 0.26 and 0.67, while for SS models, the adjusted R^2 values were higher, with ranges between 0.56 and 0.79 (Supp. Tables 2–19). The overall differences in the performance between the DIN and SS models (i.e. the generally better performance of the SS models compared with the DIN models) can be explained by the factors known to control SS and DIN dynamics in river systems (Leigh et al., 2019); this is explored

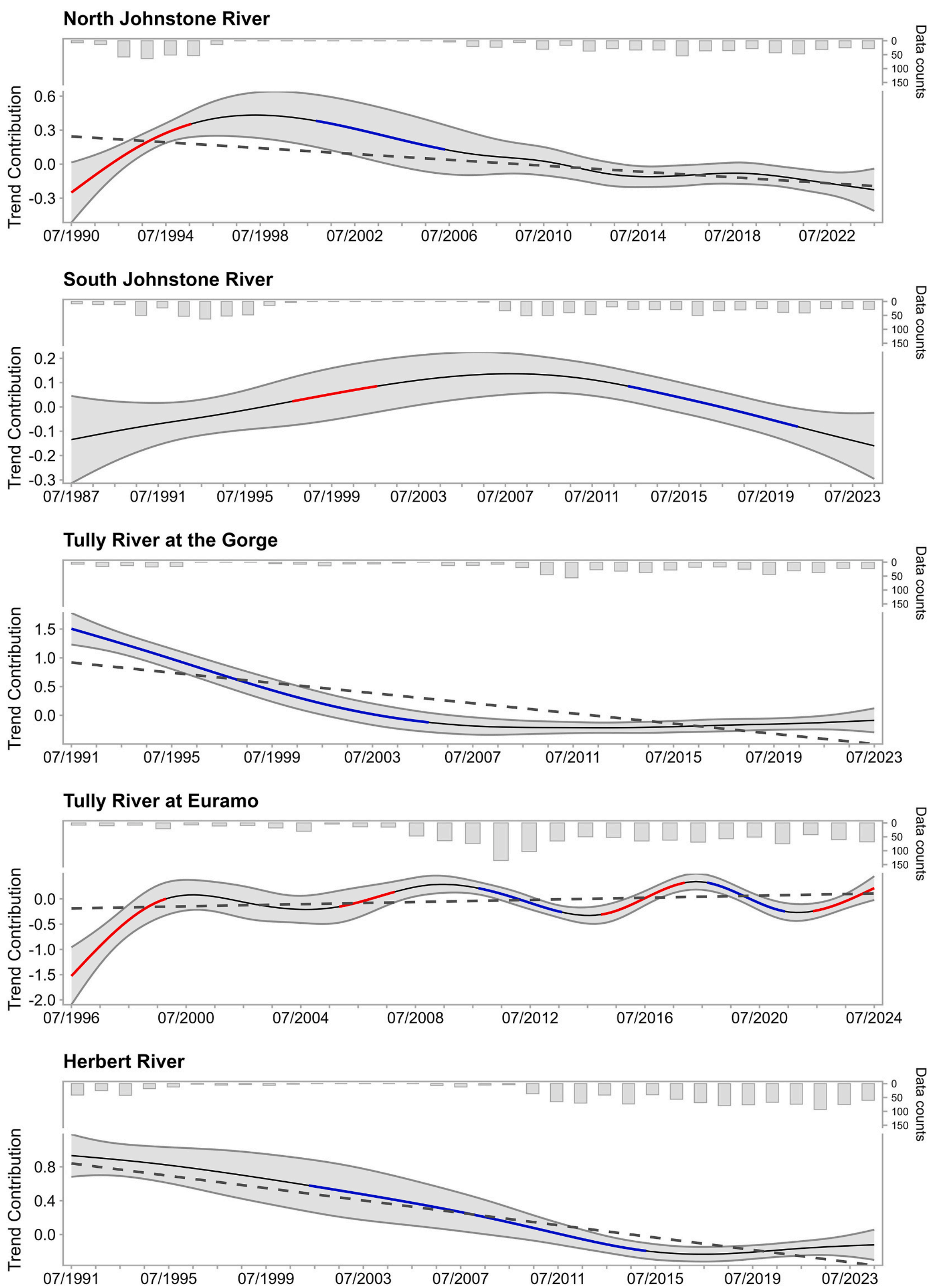

Fig. 4. Non-linear and linear trends in suspended sediment across the nine sites. Significant (episodic) periods are represented as red lines (increasing) and blue lines (decreasing). Significant linear trends are also plotted as dashed line. The hanging bar chart provides concentration data sample numbers for each fiscal year.

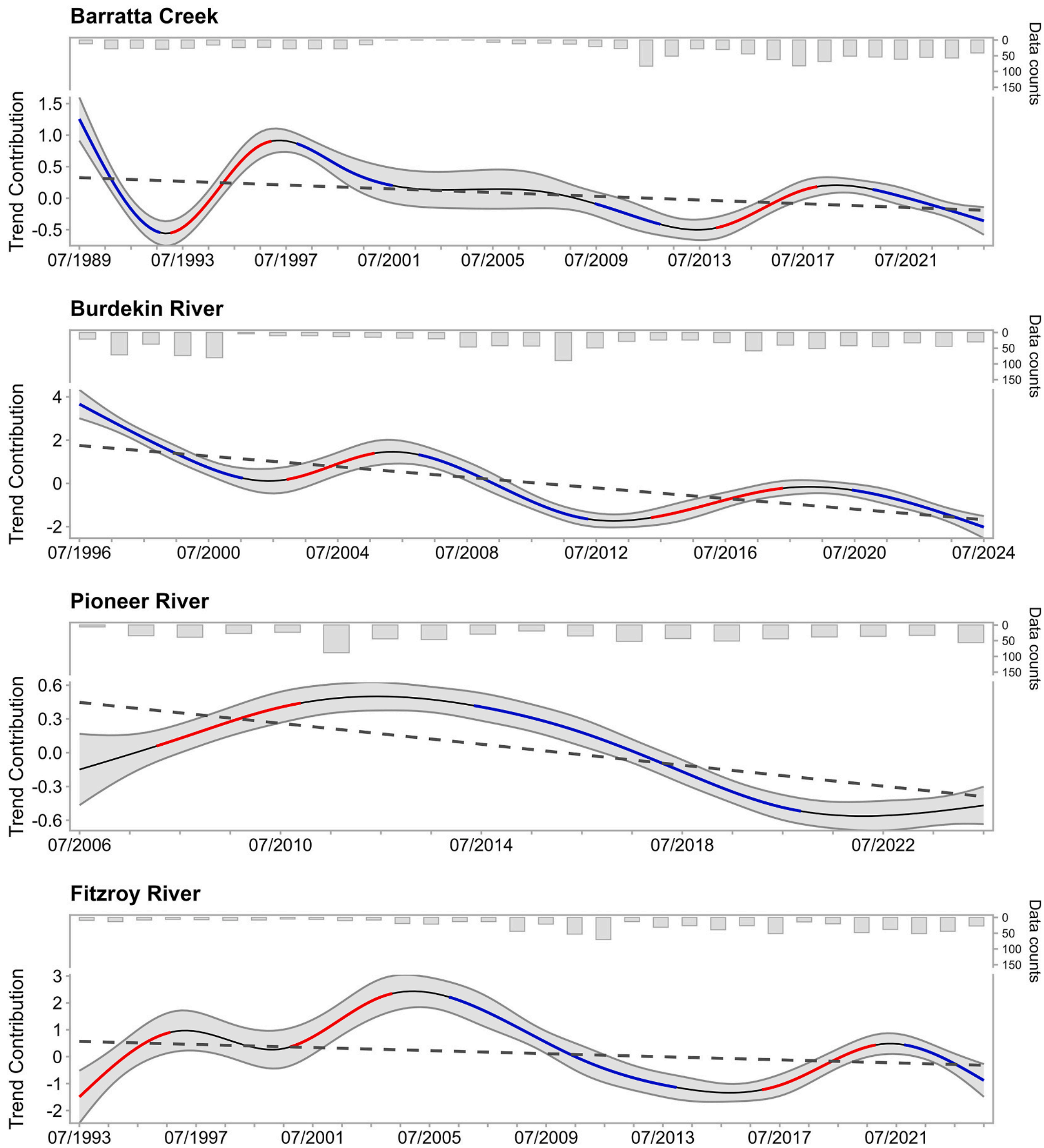

Fig. 4. (*continued*).

further in the Discussion.

For all the models developed, strong temporal dependencies (autocorrelation) were detected and accounted for in the final model structures. Multiple explanatory terms were included in all the final models; however, the specific combinations of terms differed across the models (Supp. Tables 2–19). As there is significant co-linearity between the terms; i.e. some terms act as surrogates for others, interpretation of individual term effects requires caution and instead, we focus on the broader impact of current and antecedent flow conditions on long term trends.

3.2. Water quality concentrations

The relative and absolute DIN and SS concentrations varied considerably across the catchment sites, including those within the same catchment (e.g. Tully) (Fig. 2). The highest median DIN concentrations for the raw concentration data were measured at Barratta (0.220 mg/L) and lowest at Tully Gorge (0.068 mg/L), with median concentrations for the other sites ranging between 0.115 and 0.206 mg/L (Table 2). For SS, the highest median concentrations were from the large dry tropical basins, Fitzroy (161 mg/L) and Burdekin (144 mg/L), while median concentrations from some wet tropical basins were as low as 12-15 mg/L, as measured at the Tully Gorge, North Johnstone and South Johnstone sites (Table 2). Although this study focuses on trends rather than absolute values, absolute concentrations remain important when interpreting statistically significant trends. For example, the absolute SS concentration value may decline from 0.5 to 0.1 mg/L and this would have a similar statistical result as a change from 500 to 100 mg/L; however, these relative reductions would have very different outcomes for ecosystems downstream.

3.3. Water quality trends

3.3.1. Trends in dissolved inorganic nitrogen (DIN)

All analysed sites exhibited non-linear trends in DIN across the periods of analysis (Table 4; Fig. 3). The shape of the non-linear trend response curves varied across sites. Most sites exhibited complex non-linear trends characterised by episodic periods of significant increasing (red lines) and decreasing (blue lines) DIN concentrations. However, both the timing and the duration of these periods differed across sites. The Tully at Euramo and Herbert River sites exhibited simpler response curves, with decreasing DIN during earlier (1990-

1992) and later time periods (2017-2024).

All sites but one exhibited significant long-term linear trends in DIN concentrations. Seven sites demonstrated significant decreasing trends in DIN concentration (North Johnstone, South Johnstone, Tully Gorge, Tully at Euramo, Herbert, Burdekin and Pioneer catchments; Fig. 3). Estimated annual percentage decreasing changes in DIN concentrations ranged from 0.43% (Tully at Euramo) to 3.0% (Pioneer) (Table 4). The majority of trends were strongly significant ($p < 0.001$), whereas the Tully at Euramo, Tully Gorge and the Pioneer sites showed comparatively weaker evidence ($p = 0.007$, $p = 0.002$ and $p = 0.003$ respectively). Barratta Creek exhibited an overall increasing trend for DIN concentrations across the full time series with the percentage annual increase in DIN estimated to be 4.7% per annum (Table 4). There is a distinct change in concentration at Barratta Creek between the earlier period of the time series (1987-1992) and after 2004 when DIN concentrations are noticeably higher (Fig. 3). The Fitzroy site did not display a significant linear trend in DIN (Fig. 3).

3.3.2. *Trends in suspended sediments (SS)*

All analysed sites exhibited non-linear trends in SS across the periods of analysis (Table 5; Fig. 4). Complex non-linear trends with both episodic periods of significant increasing (red lines) and decreasing (blue lines) SS concentrations were observed for seven sites (North Johnstone, South Johnstone, Tully at Euramo, Barratta, Burdekin, Pioneer and Fitzroy). The Tully Gorge and Herbert sites exhibited much simpler generally monotonic trends of decreasing SS concentrations over the time series.

Seven sites demonstrated significant decreasing linear trends across the time series (North Johnstone, Tully Gorge, Herbert, Barratta, Burdekin, Pioneer and Fitzroy catchments), with the percentage annual decreases in SS ranging from 1.3% (Fitzroy) to 6.3% (Tully Gorge). The non-linear trend analysis for the South Johnstone site also shows a trajectory of decreasing SS concentration over the past decade, although the linear trend is not statistically significant. Tully at Euramo had a weakly significant increasing linear trend ($p = 0.045$).

4. Discussion

This study is the first of its kind to assess non-linear long-term trends in DIN and SS concentrations across nine water quality monitoring sites in the catchments draining to the GBR, encompassing tropical river systems characterised by high hydrological variability and substantial differences in catchment area, land use, climate, and river regulation. Collectively these sites represent ~66% of the GBR catchment area, with data records spanning up to ~37 years. GAMMs were used to estimate temporal trends in constituent concentrations identifying non-linear trends at all sites, some of which were quite complex. Significant linear trends were also detected at many sites. GAMMs provide a flexible but statistically robust framework for analysing environmental data time series (Simpson, 2018). This approach can quantify the magnitude of change in constituents whilst accommodating both autocorrelation between samples and irregular sampling intervals. GAMMs also provide an opportunity to include a range of explanatory variables that extend beyond hydrological predictors to other potentially important drivers of constituent concentrations. The visual representation of both linear and non-linear trends serves as a useful communication tool, and the inclusion of uncertainty estimates enhances transparency and conveys the level of confidence in each model.

While a strong focus on statistical methodology for trend analysis is fundamental, it is not, on its own, adequate; consideration must also be given to the characteristics and limitations of the input data (i.e. understanding data provenance, structure, and limitations). The assessment of data suitability for inclusion in statistical models is essential and requires a cross-disciplinary team to ensure that the analysis is fit-for-purpose (Hartman et al., 2025). This study explicitly considers the presence of bias in the underlying datasets, recognising that, without substantial data exclusions, such bias cannot be completely removed. In particular, more recent monitoring data are biased towards high-flow periods, while earlier data are more representative of low-flow conditions. Given that constituent concentrations typically increase with flow, these combined biases would be expected to produce an apparent increasing trend over time. However, for most sites the contrary occurred, and we therefore cautiously conclude that the detected trends are unlikely to be artefacts of changing sampling bias alone and instead reflect genuine long-term changes.

4.1. *Use of linear and non-linear statistical approaches for water quality trend detection*

Linear trends derived from the full time series summarise the direction and magnitude of long-term change across the entire measurement record. Although most sites exhibited significant linear trends, a number of sites also demonstrated complex, non-linear (episodic) responses, characterised by alternating periods of increasing and decreasing trends throughout the study period. This highlights that a focus on simple monotonic trends can overlook the complex inter-annual and, in some cases, decadal variability arising from natural and anthropogenic climate-driven rainfall–runoff processes. Non-linear analyses provide additional insight by identifying shorter-term periods of change and potential change points and demonstrate that trend estimates can be sensitive to record length and the choice of start and end years (Burt et al., 2010). Although similar approaches have been applied elsewhere, their application to large, climatically variable tropical river systems remains limited particularly where the emphasis is on identifying long-term trends rather than underlying processes. These non-linear patterns may be more relevant when trying to understand how external factors like weather/climate, cyclones and land management interventions influence water quality through time. Additional ancillary data that quantifies changes in land use and land management are required to help interpret both the linear and non-linear responses.

4.2. *The influence of site characteristics and spatial and temporal scales*

The nine sites in our trend analysis have large differences in catchment size, climate variability and areas of cropping and grazing in their respective upstream catchments (Table 1). These characteristics greatly influence the ability to detect trends in water quality resulting from management interventions and serve to emphasise the importance of choosing suitable monitoring locations that preferably represent a single land use. Even within a single land use, large differences in soil type, slope, vegetation and management, can make detecting change challenging even at small catchment (~10 km^2) scales (as noted in Bartley et al., 2012). For example, the large dry tropical catchments of the Burdekin and Fitzroy have extreme inter- and intra-annual flow variability (Puignou Lopez et al., 2025), highly variable SS concentrations across different contributing sub-catchments (Packett et al., 2009; Bainbridge et al., 2014) and high variability in ground cover across wetter and drier years (Kuhnert et al., 2012). Given this complexity, the ability to detect trends and attribute them with confidence to specific upstream management interventions is limited at larger spatial scales.

Challenges in detecting water quality changes resulting from agricultural land management have been well documented in other studies. For example, Melland et al. (2018) showed that while positive effects on water quality may emerge within 1 to 10 years following management interventions, the time required to confidently detect such effects typically ranged from 4 to 20 years, with response times generally increasing with catchment size. Kroon et al. (2014) highlighted that in many cases, land management changes must be substantial to achieve significant pollution reductions and improve marine ecological outcomes. Other studies have shown that although soil conservation programs can result in reduced sediment loads at local scales, trapping of sediment by dams represents the dominant cause of reduced sediment loads at the global

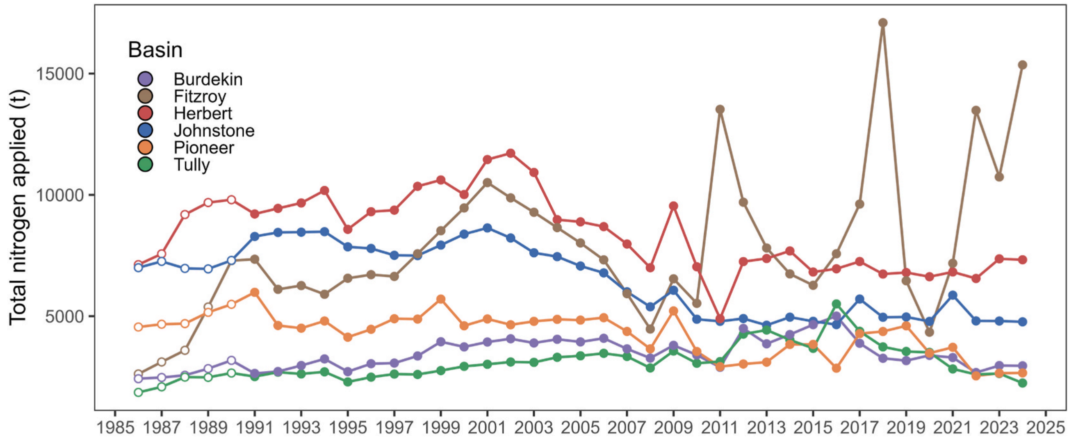

Fig. 5. Annual total synthetic nitrogen fertiliser application time series (1986 to 2024) for the river basins associated with the monitoring sites (open circles – Pulsford (1996) and closed circles – Lewis et al., In press).

scale (Syvitski et al., 2005; Walling, 2006).

Lindenmayer et al. (2022) outlined the key requirements for monitoring programs attempting to detect effects from environment management interventions, including the need for a proper experimental design. Indeed, changes in water quality following land use change (from natural to cropping or to grazing) have been detected in GBR river catchments at smaller spatial scales (<20 ha) using a BACI (Before-After-Control-Impact) sampling framework (Elledge and Thornton, 2017), and long-term (>20 year) studies employing BACI designs have successfully quantified the effects of varying grazing intensities on SS concentrations (Thornton and Elledge, 2021). Management interventions such as locally targeted gully restoration can detect water quality improvements in relatively short timeframes (~2-3 years) if the treatments cover a large proportion (>25%) of the monitored site (see Hawdon et al., 2025; Bartley et al., 2025). However, these individual sites often represent < 0.01% of the total SS load being delivered from the entire basin, so unless a very large number of sites are being treated within a catchment, detecting statistically significant changes to end of catchment SS concentrations and/or loads will be challenging. Long-term changes in vegetative ground cover may conceivably be linked to reductions in SS concentrations; however, a thorough analysis of ground cover trends was beyond the scope of this study. Importantly, reliance on ground cover metrics alone to infer changes in grazing land management may overlook additional drivers of vegetation condition,

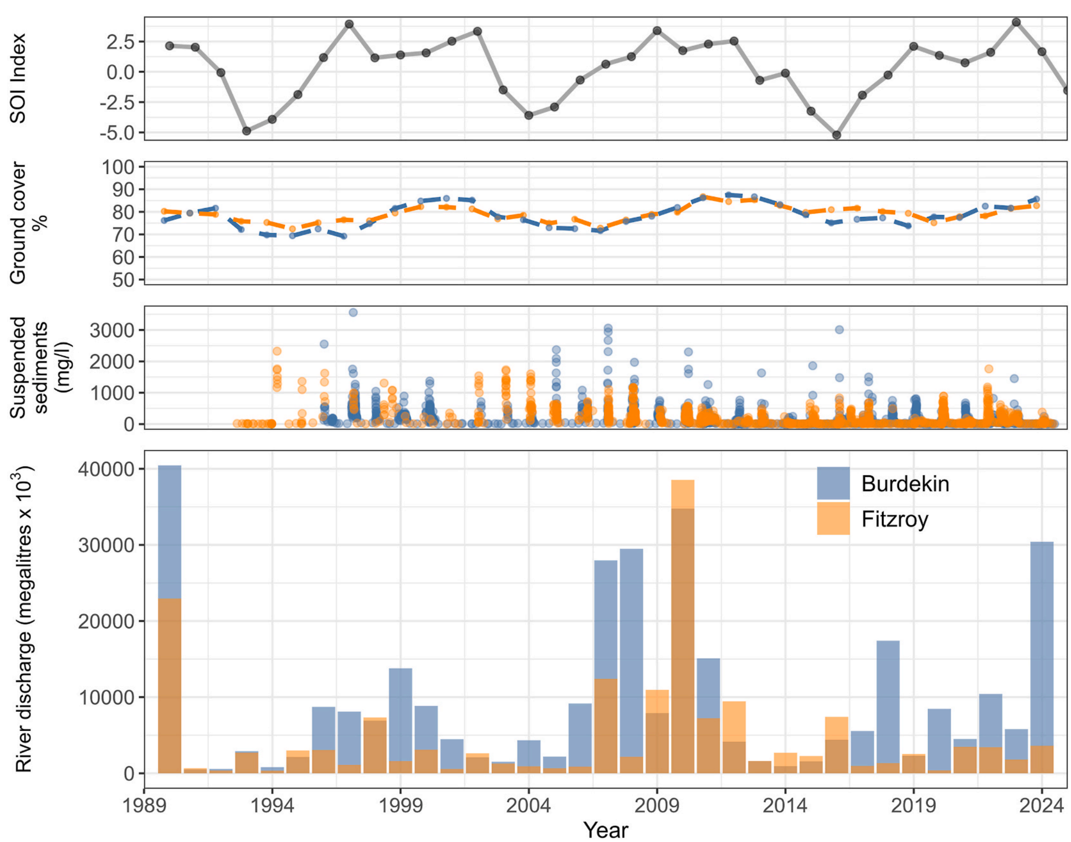

Fig. 6. Annual river discharge (fiscal year) for the Burdekin River (120006B - Clare) and Fitzroy River (130005A – The Gap), also showing the five-yearly running average Southern Oscillation Index (SOI) values and percent dry season ground cover for the Burdekin and Fitzroy basins (Sources: DLGW&V, 2025; Bureau of Meteorology, 2024; Department of the Environment, T. & Joint Remote Sensing Research Program, 2022).

runoff, and water quality, particularly shifts in grass species composition and the consequent changes in ecohydrological functioning (see Stokes et al., 2023).

While most of the sites analysed in this study do not meet the 'ideal criteria' described above to quantify water quality trends over time due to management intervention, they are the only locations where long-term (>20 years) water quality data exist across the GBR catchment area. Importantly, all the monitoring sites are located within the key basins where targets for DIN and/or SS load reductions have been established to improve water quality for the GBR lagoon (Waterhouse et al., 2026). Those sites located within the Burdekin and Fitzroy catchments effectively capture their entire basin and hence provide a direct measure of the water quality trend. Other sites, such as that in the Herbert catchment, capture substantial upstream sugarcane cropping areas (24,600 ha). Establishing new monitoring locations to evaluate water quality trends is impractical, given management targets are expected to be achieved within the next decade (Waterhouse et al., 2026). In this regard, we need to utilise the datasets immediately available, acknowledge their limitations, and draw on multiple lines of evidence to test the validity of the trends identified in our analysis. The following sections provide a critical examination of the plausibility of the trends identified in our analysis within the context of other relevant datasets including fertiliser application records, climate variability and documented management interventions.

4.3. Interpreting water quality DIN trends in the context of land use

Significant decreasing DIN trends were detected at seven sites: North Johnstone, South Johnstone, Tully Gorge, Tully at Euramo, Herbert, Burdekin and Pioneer (Table 4; Fig. 3). The timing of observed reductions in DIN concentrations for the North Johnstone, Tully at Euramo, Herbert, Burdekin and Pioneer sites broadly aligns with reductions in synthetic nitrogen fertiliser application within each of these basins (Fig. 5; Lewis et al., In press). There are, however, several confounding factors that influence stream DIN concentrations, including upstream flow sources draining different vegetation types, water storages in upstream reaches, inter-annual variability in discharge, and impacts associated with tropical cyclones. Hence, the declining DIN concentrations may be explained by several different processes (or a combination of processes) that cannot be easily disentangled.

Reductions in synthetic nitrogen fertiliser application to crops occurred in the early to mid-2000s in the Johnstone and Herbert basins, and more recently, over the past decade, in the Tully, Burdekin, and Pioneer basins (Fig. 5; Lewis et al., In press). DIN concentrations in GBR streams display a strong relationship with the absolute upstream area of fertilised cropping (Mitchell et al., 2009). Even streams with a relatively small proportion of fertilised cropping (i.e. 1% of the basin) exhibit notably higher DIN concentrations compared to streams with no cropping (Bainbridge et al., 2009). These observations indicate that fertiliser application in the upstream catchment area is a dominant influence on DIN concentrations in streams within the GBR river catchments.

However, there are several factors that affect our confidence in solely attributing these trends to fertiliser management, including the extent to which the monitoring sites represent catchment-wide fertiliser trends and the proportion of fertilised area in the upstream reaches. Where a large proportion of the cropped area lies upstream of the site (e.g. the Tully at Euramo site captures 75% of the total Tully basin cropping area), we have a higher degree of confidence that the recent decline in DIN concentrations can be attributed to decreasing trends in nitrogen fertiliser application (Fig. 5; Lewis et al., In press). In contrast, the North and South Johnstone sites capture only a small proportion of the total Johnstone basin cropping area (13% and 6%, respectively). Consequently, we have lower confidence that the broad patterns in nitrogen fertiliser usage in this basin are adequately reflected in the DIN trends observed at these two monitoring locations.

In some instances, declines in DIN concentrations do not directly correspond with reduced nitrogen fertiliser application. Over the past decades, the sugarcane and banana industries have adopted improved/new farming practices that are expected to reduce DIN losses at the catchment scale, independent of total fertiliser inputs. These include improved fertiliser placement or product choice (Thorburn et al., 2024), implementation of nutrient management programs such as Six Easy Steps and Soil specific nutrient management guidelines (Schroeder et al., 2010) and, addressing agronomic constraints that improve fertiliser uptake. Together these changes may help explain the observed declines in DIN concentrations in several catchments including the Herbert, Tully at Euramo, Burdekin and Pioneer over the past decade (Fig. 3). Declining DIN trends at the Tully Gorge site, which captures no upstream area of fertilised cropping, highlights the need for caution when attributing reductions in DIN concentrations solely to decreased fertiliser application, without first considering broader land use or management changes occurring upstream. Changing stream DIN concentrations may also be explained by interactions with SS concentrations. For example, Garzon-Garcia et al. (2024) found particulate nutrients associated with SS can be rapidly transformed into DIN during instream transport. Hence, it is possible that some of the declines in DIN concentrations are associated with concurrent declining trends in SS, as observed at several sites (i.e. North Johnstone, Tully Gorge, Herbert, Burdekin and Pioneer).

Strong evidence for a relationship between fertiliser use and DIN concentrations is found at the Barratta site, where DIN exhibited a significant increasing linear trend. Furrow-irrigated crops (predominantly sugarcane) were established in this catchment following the construction of the Burdekin Falls Dam in 1987, which supplied the irrigation water for crop expansion. Cropping in Barratta Creek (introduced in the early 1990's) is relatively recent compared to other catchments where cropping was established by the 1920's (Lewis et al., 2021). Cropping represents ~19% of the catchment area of the Barratta Creek monitored site, which is comparable to the Pioneer site, but higher than that of the other study sites (Table 1). This analysis demonstrates that major changes in Barratta DIN concentrations occurred during the mid-1990s, coinciding with the conversion of upstream areas from grazing lands to fertilised sugarcane farms. Hence, the increase in DIN concentrations at this site can be clearly linked to upstream land use change and increasing use of nitrogen fertilisers. However, when we only focus on the current decade, DIN concentrations at Barratta show no clear trend (Fig. 3), highlighting the importance of incorporating long-term datasets and carefully selecting the temporal window for trend analysis.

The Fitzroy site did not exhibit a significant linear trend in DIN (Fig. 3), although non-linear patterns were evident. These included a significant period of decreasing DIN concentrations following a wetter period of increased cyclone activity during the early 2010's (e.g. the extreme 2010/2011 event associated with ex-Tropical Cyclone Tasha; Fig. 6), followed by a more recent increase during a drier period. This latter increase also coincides with increased nitrogen fertiliser application by both grains and an expanding cotton industry within this basin (Fig. 5; Lewis et al., In press; Lewis et al., 2021).

4.4. Interpreting water quality suspended sediment trends in the context of land use and condition

Significant decreasing SS trends were detected at seven sites: North Johnstone, Tully Gorge, Herbert, Barratta, Burdekin, Fitzroy and Pioneer (Table 5; Fig. 4). The Fitzroy, Burdekin and Herbert sites represent three of the largest GBR basins and collectively contribute 62% of the SS load delivered by the GBR catchment area to the lagoon (Fig. 1; McCloskey et al., 2021a). All three basins were found to have significant decreasing SS trends, with percentage decreases per annum ranging from 1.3% for the Fitzroy to 4.5% for the Burdekin (Table 5). These large basins with lower and more variable rainfall, are particularly influenced by wet-dry climatic cycles (Fig. 6). This results in more variable ground cover and hence SS concentrations tend to be higher during (or following) periods of drought and, lower when the basins receive above average rainfall (Kuhnert et al., 2012; Bainbridge et al., 2014). The discharge from these rivers closely follow the El Niño Southern Oscillation Index (SOI; Fig. 6) and the SS concentration trends broadly mirror this pattern, with periods of elevated discharge associated with La Niña events coinciding with lower SS concentrations. This finding indicates that these rivers are sediment supply limited, meaning there is more energy than sediment available, which is consistent with previous research by Amos et al. (2004) for the Burdekin River. This also suggests that if rainfall and runoff increase with climate change, then more sediment may be delivered in the periods of drought-breaking events which occur during lower ground cover (McCulloch et al., 2003).

Long-term landscape repair remediation programs have been implemented in some, but not all, the river basins explored here. These initiatives are designed to improve management of ground cover on grazing lands and include targeted investments to remediate gully and streambank erosion, with a key objective of reducing SS loads. However, these land management improvements are difficult to untangle from the broader climatic signal, particularly during recent wetter conditions. At the Burdekin site (Figs. 4 and 6), declines in SS concentrations occurred in the late 1990s, late 2000s, and early 2020s, broadly coinciding with wetter periods that followed extensive drought in the catchment (early to mid-1990s and early 2000s). Similar long-term patterns are evident at the neighbouring Fitzroy site, with pronounced periods of increasing and decreasing concentrations, likely reflecting variability in overall rainfall and runoff in this basin (Figs. 4 and 6). The most recent declines over the past ~ five years may reflect the cumulative effects of improved grazing practices and gully and streambank remediation; however, attributing these changes to management interventions remains challenging given the strong influence of concurrent climatic variability.

The declining Herbert SS trends started at the beginning of the time series in the early 1990's, coinciding with potentially influential land use changes that occurred in the late 1980's across both the upper and lower Herbert catchment (Fig. 4). These include the collapse of the tin mining industry in the Herberton area (upper Herbert) (Little, 2014; unpublished report), the closure of the logging industry due to World Heritage Listing (Vanclay, 1996) and the rapid and widespread adoption of Green Cane Trash Blanketing by the sugarcane industry in the lower Herbert (Wood, 1991). The adoption of Green Cane Trash Blanketing is known to greatly reduce sediment erosion rates from cropping lands (Prove et al., 1995) and will be an influential factor on SS trends from the late 1980s to early 1990s on neighbouring Wet Tropics basins and other rainfed sugarcane systems.

Regardless of the processes driving the declining trends in SS for these larger rivers, the upstream catchments appear to be in better condition in terms of seasonally adjusted ground cover (Bastin et al., 2014). However, ground cover does not always correlate with grass biomass or land condition (Stokes et al., 2023) and tree clearing is still occurring in some areas (Queensland Government, 2025). These factors can have complex and interacting effects on stream baseflow and event flow dynamics (Peña-Arancibia et al., 2012; Cheng and Yu, 2019). Actively eroding gullies and streambanks also remain widespread and are the dominant source of erosion in many grazed river basins of northeastern Australia (Wilkinson et al., 2013). Reduced rates of floodplain deposition since the mid-twentieth century at sites previously dominated by gully erosion suggest that sediment derived from gully networks may be declining in some river basins (e.g. Fitzroy) (Hughes et al., 2009) or stored in alluvial bench features within the stream network (Pietsch et al., 2015). Given that landscape recovery typically occurs far more slowly than degradation, the declining SS concentrations observed at most sites are nevertheless a promising sign.

Analysis of Barratta Creek indicates that the conversion of grazing land to sugarcane cropping during the early 1990s initially coincided with a period of increased SS concentrations and hence likely linked to runoff from newly cleared bare paddocks. However, over the subsequent two decades, SS concentrations have remained relatively stable under continued sugarcane cultivation (Fig. 4). This finding suggests that sediment erosion is less of a problem for this flat lower Burdekin floodplain landscape. In contrast, the decreasing SS concentrations in the North Johnstone, Tully Gorge and Pioneer sites are more difficult to attribute and may reflect the influences of multiple factors (see also Bahadori et al., 2020). All three sites are in wetter parts of the GBR catchment area, and their upstream catchments contribute low SS concentrations relative to the larger drier basins (Fig. 1). Hence a decrease in concentration over the time series will reflect a relatively minor change in absolute concentration (e.g. if the fitted trend at Tully Gorge was sustained, the median starting concentration of 12 mg/L would be expected to fall to <5 mg/L over the full 32-year time series; Table 5). There have been focused streambank and streambed restoration projects along several of these rivers (e.g. Pioneer); however, it is difficult to directly link these activities to declines in SS. Furthermore, some of these trends may be related to lag effects associated with weir/dam construction in their upstream areas (Table 1). With time, channels downstream of dams may have stabilised or contracted, reducing sediment delivery to downstream areas. Both the North Johnstone and Tully Gorge sites are also likely to have been influenced by the closure of the logging industry following World Heritage listing in 1988 (Vanclay, 1996), although the effects on stream SS concentrations due to lag times from when logging ceased are not well known. The area upstream of the Tully Gorge site is 90% forested with around 34% of its runoff captured by Koombooloomba Dam (Table 1). This SS decline may also reflect other non-agricultural related issues such as a change in the number and frequency of landslides or cyclone impacts (Nott, 2018).

In contrast to the other study sites, a weakly significant increasing SS trend was detected at the Tully at Euramo site. This time series displayed a complex non-linear trend with significant periods of both increasing and decreasing concentrations, all occurring within a comparatively low median SS concentration and range (25 ± 38 mg/L) relative to the other study sites (Fig. 2, Table 2). Although this time series encompassed significant land use changes within this basin, it is difficult to attribute specific climatic drivers or land management changes to these episodic trend periods given the relatively low magnitude of change and the variability inherent in the system. Major land use changes include the conversion of coastal grazing to sugarcane cultivation (1980-1990's), the expansion of the banana industry in the 2010's following Tropical Cyclone Yasi (Lewis et al., 2021) and the recent removal of several banana plantations (from ~2020) due to Panama disease.

4.5. *Comparison with other studies in the GBR*

There have been several other recent studies evaluating long-term trends in SS and the closely related turbidity concentrations for the GBR river catchments (Guo et al., 2025; Guo and Hairsine, 2025). Using the WRTDS model, Guo et al. (2025) reported increases in SS for five sites over the 2000-2020 period within the northeast region of Australia which included two of the sites studied here (Tully River at Euramo, Barratta Creek at Northcote; D. Guo, pers. comm., 03/03/25). Although our analysis did find a weakly significant increasing SS trend for the Tully River at Euramo site (1996 to 2024), contrary to the Guo et al. (2025) study, we found a significant decreasing SS trend at Barratta Creek (1989-2024). Guo and Hairsine (2025) reported a general upward trend in turbidity at the Fitzroy and Herbert River sites, and a downward trend at the Burdekin River (at Clare) over varying time periods ranging between 13 and 41 years. In this study we found a decrease in SS at all three sites for periods from 28 to 36 years. Although the datasets used in this analysis share some common elements with these previously discussed studies, such as the inclusion of the GBRCLMP data for the 2010 to 2019 period (e.g. Guo et al., 2025), there are many data and methodological differences that need to be considered when comparing these results. These include the differences in data types (SS versus turbidity and, modelled flow versus measured river flows), data sources (notably extensive earlier data described in James et al. (2025) were used in our study that were not available at the time of this earlier publication), treatment of outliers, overall time periods studied and difference in modelling methods. Beyond these technical differences, the WRTDS model was applied by Guo et al. (2025) using uniform default parameterisation across 287 monitoring sites nationally, with study period and flow inputs selected to optimise national coverage rather than regional specificity. The framework is therefore designed to identify broad directional trends at continental scale, with regional patterns interpreted as secondary findings. In contrast, the GAMM framework employed here was developed specifically for the highly seasonal and episodic flow regimes of GBR catchments, with a carefully curated regional dataset. Discrepancies in results are therefore not unexpected and likely reflect the different primary objectives of the two approaches rather than errors in either.

Other studies have also developed statistical water quality models using some of the same datasets analysed here; however, these studies primarily focused on understanding drivers of water quality variability (e.g. Liu et al., 2018, 2021) or on forecasting and prediction (e.g. Khan et al., 2020). For example, using an 11-year concentration dataset spanning 2006 to 2016, Liu et al. (2018, 2021) found that, for most constituents, natural catchment characteristics such as climate, topography, geology and hydrology were more influential than anthropogenic landscape factors in explaining spatial variations in water quality. However, they also found a positive correlation between NO_x (oxidised nitrogen) and the proportion of area occupied by sugarcane. As noted by Liu et al. (2021), there may be cross-correlation between natural and anthropogenic drivers, as many agricultural activities are strongly influenced by climate, reducing the ability to detect an anthropogenic water quality signal. While these studies provide important insights into the drivers of water quality patterns, they differ from the present study, which focuses on quantifying and interpreting long-term trends and periods of change in water quality.

4.6. Considerations for future water quality monitoring programs, implications for management and areas of further research

In the GBR catchment area, water quality monitoring programs have several key purposes including supporting marine modelling efforts (Steven et al., 2019) and calibrating catchment models (McCloskey et al. 2021a, 2021b). Results from this study cannot be directly compared with modelled water quality load predictions (McCloskey et al. 2021a, 2021b) or to the Reef Report Card data (https://www.reefplan.qld.gov.au/tracking-progress/reef-report-card) as these programs present constituent loads and not concentrations. Resourcing for this study did not allow for a rigorous evaluation of loads; however, this comprehensive evaluation of DIN and SS concentration data now provides a robust foundation for integrating constituent concentrations with rainfall and runoff data. Future development of trend analyses for constituent loads will further strengthen the evidence base and support for the prioritisation of catchment management actions.

This study has also demonstrated that consistent and robust water quality data are required for quantifying trends in water quality concentrations delivered to the GBR. Attributing the underlying causes of observed trends in large, mixed land-use river basins remains challenging, and will require a combination of (i) well designed studies at smaller spatial scales; and (ii) additional supporting evidence such as spatial and temporal data on nitrogen fertiliser application amounts, changes in land use and management and a publicly available inventory of remediation activities. The use of multiple lines of evidence, and multiple explanatory data sources, will improve the confidence with which we can interpret and communicate water quality trends (see Bainbridge et al., 2023). There are several additional sites within the GBRCLMP that now approach 15 plus years of continuous water quality monitoring, and they may soon also provide a suitable dataset to analyse long-term water quality trends. Future research efforts could also analyse trends for other important water quality parameters such as particulate nitrogen, particulate phosphorus and dissolved inorganic phosphorus.

Many of the major land use changes in the GBR study catchments occurred prior to the collection of the first water samples used in this study (c. 1986). This study did not assess changes in concentrations relative to predevelopment conditions - such analyses require calibrated catchment models or investigations based on long half-life isotopes. Other studies have shown that constituent loads in the GBR are likely to be between one and five times greater than pre-development conditions (e.g. Mariotti et al., 2021; McCloskey et al., 2021a,b). Importantly, as many of the monitoring sites are located well upstream of the 'end of catchment', declining concentrations measured at the site may not always translate to declining concentrations in the adjacent inshore marine environment due to legacy effects including the resuspension and remineralisation of sediments and attached nutrients. Ecologically sustainable water quality concentrations and exceedance values also vary between freshwater and marine ecosystems. Therefore, despite DIN and SS concentrations showing positive downward trends for most sites, further analysis is required to determine if these reductions translate to corresponding improvements in marine water quality (Gruber et al., 2026). Improved understanding of catchment-to-marine linkages based on empirical relationships would provide a clear indication that management actions are resulting in improved water quality in the GBR lagoon; however, until such analyses are undertaken, these interpretations should be treated with caution.

Future research should now focus on combining DIN and SS concentrations with stream discharge to determine trends in constituent loads over time. This could also include evaluating trends under different flow conditions (event vs. baseflow) as well as considering the spatial variability in river catchment runoff, as this may help reveal key pathways and underlying drivers of water quality change (e.g. Zhang et al., 2021). The impact of overbank flows, a frequent occurrence for many of these rivers, would also need to be addressed in any future evaluation of load trends (Wallace et al., 2012). An evaluation of changes in constituent loads could also incorporate projections of future water quality loads based on climate change scenarios. Although no significant trends in mean annual total streamflow have been detected for most rivers draining into the GBR up to 2021 (Wasko et al., 2024; but see Lough et al., 2015), other hydrological metrics may nevertheless be changing, such as rainfall intensity, peak discharges and flood travel times (see Luo et al., 2026). Changes in vegetation structure such as increases in foliage projective cover and woody stem density in some land uses, driven by rising atmospheric carbon dioxide, may also be occurring in parts of these catchments (Crowley and Murphy, 2023). Such changes could influence upstream ground cover and catchment hydrology. Progressing this analysis of constituent concentrations to an analysis of constituent loads is important because, even at sites showing statistically significant declines in concentrations, future changes in climate variability such as more extreme droughts and floods could, and possibly already are, resulting in higher overall consistent loads entering the GBR (e.g. Lough et al., 2015). That is, any improvements in land management, if detectable, may be offset by effects associated with climate change. This would necessitate sustained or greater emphasis on maintaining adequate vegetation cover across all areas, in addition to ongoing improvements in fertiliser and soil management.

5. Conclusion

This study applied GAMMs to quantify long-term (~linear) and episodic (non-linear) trends in water quality across nine sites within the Great Barrier Reef catchment area. Seven of the nine sites exhibited significant long-term decreases in DIN, SS or both. Importantly, all sites displayed significant episodic trends in both DIN and SS concentrations, providing additional insights into the timing and magnitude of change throughout the monitoring record. These temporal patterns can be compared with explanatory catchment datasets to help identify potential drivers of change. Encouragingly, our analysis indicates declining trends in the most recent record of several monitoring sites which may be associated with reduced fertiliser application, improved catchment condition and/or management interventions. Overall, this study indicates that stream water quality responds to upstream catchment changes through complex processes. Although the underlying causes of these changes are not yet fully resolved, the trends are moving in the right direction. Further work is needed to evaluate whether these concentration patterns are also evident in constituent loads, which will be increasingly important under current and future climate change.

Our analysis provides insights on the challenges to both measure and meet water quality targets across the Great Barrier Reef catchment area, and the associated challenges in detecting long-term water quality changes. This study serves to highlight the realities of available water quality monitoring data which are often biased to certain flow conditions, represent non-continuous time series and do not always fully capture the land use or management change in their upstream reaches. In this regard, analysts of such datasets need to develop innovative ways to overcome the limitations and challenges this poses. Approaches such as isolating flow sources to the upstream catchment area where management has occurred, coupled with a multiple lines of evidence framework including independent modelling, fertiliser application history, ground cover change, and management interventions offer greater confidence and support for, the attribution of observed trends at the sampling sites.

CRediT authorship contribution statement

Cassandra S. James: Conceptualization, Data curation, Formal analysis, Methodology, Visualization, Writing – original draft, Writing – review & editing. **Petra M. Kuhnert:** Formal analysis, Methodology, Validation, Visualization, Writing – original draft, Writing – review & editing. **Stephen E. Lewis:** Conceptualization, Data curation, Methodology, Writing – original draft, Writing – review & editing. **Rebecca Bartley:** Conceptualization, Funding acquisition, Methodology, Project administration, Writing – original draft, Writing – review & editing. **Emma Lawrence:** Formal analysis, Methodology, Writing – review & editing. **Jennifer Strauss:** Formal analysis, Methodology, Writing – review & editing. **Reinier M. Mann:** Writing – review & editing. **Ryan D.R. Turner:** Writing – review & editing. **Zoe T. Bainbridge:** Conceptualization, Data curation, Funding acquisition, Methodology, Project administration, Writing – original draft, Writing – review & editing.

Declaration of competing interest

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

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2026.130359.

Data availability

Water quality data are available on the Queensland state government's Tahbil Water Quality Data Portal and Water Monitoring Information Portal. Discharge data is also available on the Water Monitoring Information Portal. Barratta Creek water quality data are available upon request.

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