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Wetland-use and wetland-cover change and ecological integrity of wetlands around Kibale National Park, Kamwenge District, Western Uganda

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ABSTRACT

Wetlands around Kibale National Park support biodiversity, regulate water quality and sustain livelihoods but face increasing anthropogenic pressures. This study assessed wetland-use and cover change (between 2016 and 2025), macrozoobenthos diversity and physico-chemical water quality in Bigodi, Kahunge and Businge wetland systems within a 5 km buffer of Kibale National Park, Kamwenge District, Uganda. Sentinel imagery was classified to determine cover change. Sampling sites were classified as high quality, medium quality and low quality using the wetland rapid assessment procedure (WRAP). These categories represented a disturbance gradient based on ecological and anthropogenic characteristics of the wetland sites. Macrozoobenthos were sampled in dry (January 2025) and wet (April 2025) seasons across all quality sites a scoop net, identified to family level and analyzed for diversity indices and average score per Taxon (ASPT). Physico-chemical parameters were measured in situ and in a laboratory. Wetland-use and wetland-cover (WUWC) change varied: Bigodi wetland system gained 1.17% wetland, Kahunge wetland system lost 3.58% and Businge wetland system lost 8.47% with +7.64% grassland increase. Macrozoobenthos abundance was highest in Bigodi (1189 individuals, 22 families), moderate in Kahunge (538, 21 families) and lowest in Businge (216, 21 families). Seasonal effects were significant in Kahunge (diversity $p = 0.002$) and Businge (diversity $p = 0.033$, Average Score Per Taxon $p = 0.048$), while ASPT differences across disturbance gradients were significant only in Businge ($p = 0.004$). Physico-chemical parameters generally met National Environment Management Authority standards, with minor nitrite exceedance in isolated low-quality sites. A principal component analysis showed system- and season-specific water chemistry: Businge dry-season linked to higher nitrates (mean = 0.413 ± 0.393 mg/L) and hardness (mean = 175 ± 144.3); Bigodi wet-season to higher Total dissolved solids (mean = 142.5 ± 80.6 ppm) and nitrates (mean = 0.675 ± 0.496 mg/L). Despite acceptable water quality, significant wetland loss, especially in Businge, threatens ecological integrity, warranting targeted habitat protection and stricter wetland-use controls.

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Introduction

Protected areas are central to global biodiversity conservation because they safeguard species, habitats and ecological processes that maintain ecosystem stability. National parks in particular are established to protect natural biodiversity together with its ecological structure and environmental processes while also promoting research, education and recreation (Stolton et al , 2013). Historically, conservation research and management have focused primarily on ecosystems located within protected area boundaries (Nash 1970; Floyd 2001; Chapman 2003; Dilsaver 2009; Katswera 2023). However, landscapes surrounding protected areas often designated for various land uses play a critical role as ecological buffers that protect biodiversity and maintain ecological connectivity (Borrini-Feyerabend and Hill 2015). Wetlands represent one of the

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most important ecosystems within these surrounding landscapes. According to the wetlands include areas of marsh, fen, peatland, or water that may be natural or artificial, permanent or temporary, with water that is static or flowing. These ecosystems perform numerous ecological and socio-economic functions, including water purification, flood regulation, nutrient cycling, carbon sequestration and habitat provision for diverse species (Schuyt 2005; Turyahabwe et al. 2013; Langan et al. 2019).

Despite their importance, global estimates indicate that more than half of the world's wetlands have disappeared during the past century due to human-induced environmental change (Mitsch and Gosselink 2007; Muyideen 2016). In Uganda, wetland ecosystems have experienced substantial decline over recent decades. National assessments indicate that wetlands covered approximately 15.6% of the country's land area in 1994 but had declined to about 10.9% by 2008 (Nsubuga et al. 2014). Conversion of wetlands to farmland, grazing areas, settlements and other land uses has significantly altered wetland ecosystems across the country. Such transformations have ecological consequences including biodiversity loss, declining water quality, reduced carbon storage and diminished ecosystem services that sustain rural livelihoods. In western Uganda, wetland degradation has been associated with vegetation clearance, drainage, crop cultivation and grazing activities that expose wetland soils to erosion and ecological disturbance (Businge and District, 2017).

The landscapes surrounding Kibale National Park in western Uganda provide a notable example of these dynamics. Kibale National Park, established in 1993, is a medium-altitude tropical moist forest covering approximately 795 km^2 and represents one of the most biologically diverse protected areas in East Africa (Hartter 2007). The park is globally recognized for hosting the largest population of habituated chimpanzees and supports numerous other species including primates, birds, amphibians and invertebrates. In addition to its ecological importance, Kibale National Park contributes significantly to Uganda's tourism sector and national economy (Obua 1997; MacKenzie et al. 2017).

However, the park exists within a highly transformed agricultural landscape characterized by rapid population growth, high population density and extensive smallholder farming (Hartter 2007). These pressures have intensified land-use change in areas surrounding the park and have placed considerable stress on natural ecosystems such as wetlands. Historical studies indicate that smallholder agriculture around Kibale National Park increased by approximately 137% between 1955 and 2000, resulting in substantial landscape transformation and habitat fragmentation (Hartter and Southworth 2009). Wetlands in the park's surrounding areas are increasingly utilized for fuelwood harvesting, construction materials, grazing and agricultural cultivation, activities that collectively threaten their ecological integrity.

Wetlands adjacent to Kibale National Park play critical ecological roles by acting as refugia, breeding grounds and ecological corridors for wildlife species associated with the park (Coops and Beklioglu, 2003). They also contribute to hydrological regulation, nutrient cycling and water purification within the broader landscape. Degradation or loss of these wetlands can therefore reduce habitat connectivity and isolate protected areas from surrounding ecosystems, potentially affecting species movement, gene flow and ecological resilience.

Assessing wetland ecological integrity requires the integration of both biological and environmental indicators. Macrozoobenthos are widely used as bioindicators in aquatic ecosystems because they respond sensitively to environmental disturbances and reflect cumulative ecological conditions over time (Tagliapietra and Sigovini 2010; Dar et al. 2019). These organisms play important roles in aquatic food webs and their community composition, diversity and abundance often change in response to pollution, habitat degradation and hydrological alteration. Consequently, macrozoobenthic communities provide reliable indicators of long-term environmental quality in wetland ecosystems.

Complementing biological indicators with water physico-chemical parameters provides insight into the short-term environmental conditions that influence aquatic life. Parameters such as pH, dissolved oxygen, electrical conductivity, total dissolved solids and nutrient concentrations are widely used to evaluate water quality and wetland ecological health (Hey et al. 1994; Reddy and Gale 1994). Variations in these parameters can indicate nutrient enrichment, contamination, or hydrological disturbance, all of which may affect biological communities within wetlands.

Despite growing awareness of wetland degradation in Uganda, relatively few studies have integrated analyzes of wetland-use and wetland-cover change with ecological indicators of wetland integrity, particularly in landscapes surrounding protected areas. Previous research has often examined either land-use change patterns or biological indicators separately, leaving important gaps in understanding how landscape transformation influences wetland ecosystem condition. This gap is particularly evident in

wetlands surrounding Kibale National Park, where limited empirical data exist linking wetland-cover dynamics with biological and water-quality indicators.

This study therefore investigated wetland-use and wetland-cover change and assessed the ecological integrity of wetlands surrounding Kibale National Park in Kamwenge District, western Uganda. Specifically, the research examined wetland-use and wetland-cover dynamics between 2016 and 2025, analyzed the spatial-temporal distribution of macrozoobenthos as indicators of habitat quality, and evaluated water physico-chemical parameters across disturbance gradients. By integrating remote sensing analysis with biological and environmental indicators, the study provides a comprehensive assessment of wetland condition within a protected area buffer landscape.

The findings provide important baseline information for wetland conservation and management around Kibale National Park. In addition, the study contributes to a broader understanding of how land-use change influences wetland ecological integrity in biodiversity-rich landscapes and offers insights relevant to policy development, ecosystem monitoring and sustainable wetland management initiatives in Uganda.

Materials and methods

Study area

The study was conducted in wetlands located within a 5 km buffer surrounding Kibale National Park (KNP) in Kamwenge District, Western Uganda. KNP covers about 795 km^2 across five districts and is bordered by small farms, tea plantations, woodland patches and wetlands (Hartter 2009). The 5-km buffer was selected because land-use pressures and ecological interactions around protected areas are most intense close to the park boundary. Only wetlands in Kamwenge District were considered to maintain consistent administrative and ecological conditions. Three wetland systems were assessed: Bigodi (5.41 km^2), Kahunge (8.75 km^2) and Businge (1.18 km^2), all situated along the park's eastern edge where anthropogenic activity is high. These wetlands are dominated by *Cyperus papyrus*, swamp forest patches, sedges and open-water channels and they provide key functions such as water purification, sediment trapping and flood regulation (Gosling 2011). The district is largely agricultural, with over 95% of residents depending on farming contributing to wetland conversion for cultivation, settlement expansion, brick-making and fuelwood extraction. The region experiences bimodal rainfall (March–May and September–November), receiving 1200–1800 mm annually, with temperatures of 18–28 °C conditions that support diverse wetland flora and fauna (Figure 1).

Sampling strategy

The study focused on wetlands within 5 km of Kibale National Park in Kamwenge District, assessing Wetland-use and wetland cover change, macrozoobenthos distribution and water physico-chemical parameters. Field sampling targeted three wetland systems; Bigodi, Kahunge and Businge-across disturbance gradients classified as high, medium and low quality using the wetland rapid assessment procedure (WRAP). Macrozoobenthos and water samples were collected in both dry (January) and wet (April) 2025 seasons to capture temporal variation.

A quantitative design was used, with qualitative identification of wetland-use/cover types and macrozoobenthos families. Sampling sites were selected based on WRAP scores, ensuring representation of intact and disturbed wetland conditions. At each site, three sampling points were purposively selected to incorporate all relevant human impacts and habitat characteristics. This strategy enabled comparative assessment of ecological integrity across space (disturbance gradients and wetland systems) and time (dry and wet seasons), providing a robust basis for evaluating wetland condition around the park.

Data collection methods

Study site selection

Sampling sites were selected using the Wetland Rapid Assessment Procedure (WRAP) (Miller 1999). When determining wetland quality using the WRAP methodology, the following six variables were

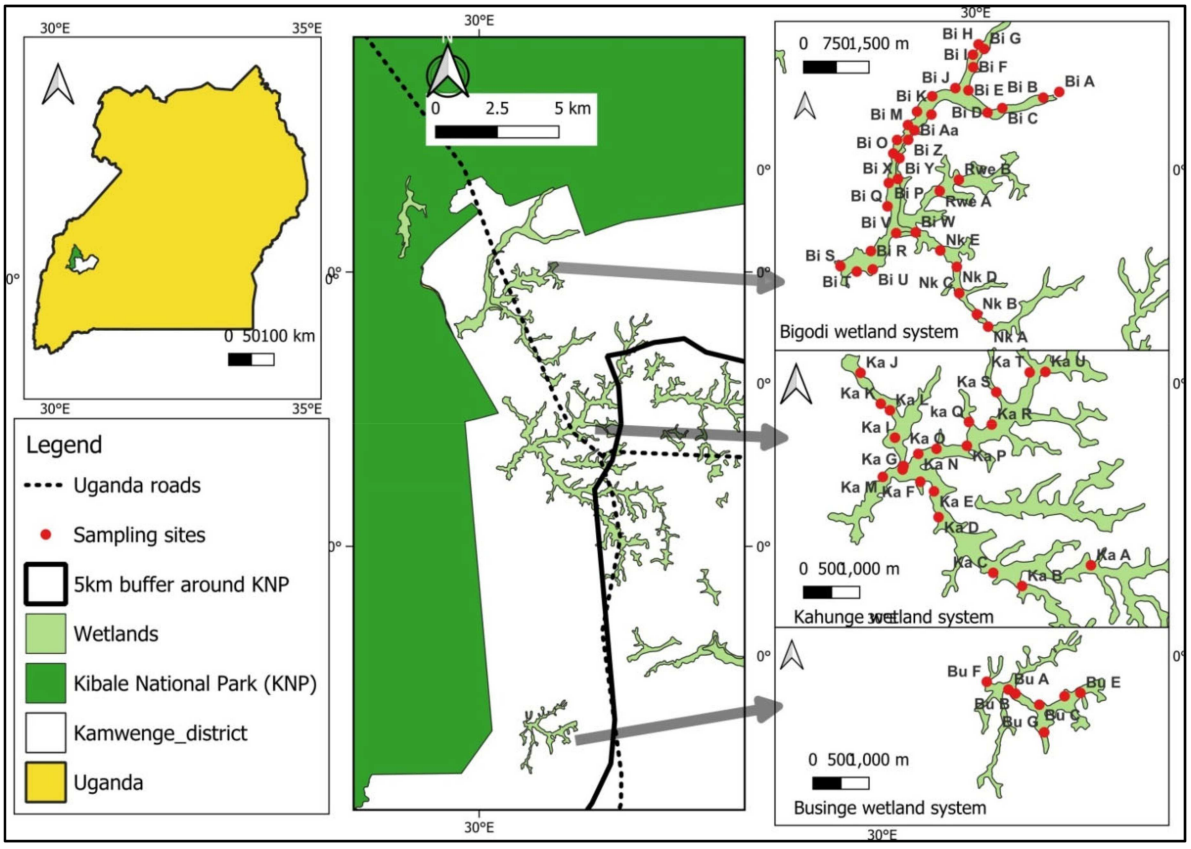


Figure 1. A map of Uganda (left), expanded (middle) is area around Kibale National Park, Kamwenge side showing the wetlands in the study area and on the left are the wetlands with sampled points (QGIS version 3.34.5-1).

assessed: wildlife utilization, wetland overstory/shrub canopy, wetland vegetative ground cover, adjacent upland support/wetland buffer, field indicators of wetland hydrology and water quality input and treatment systems. For each wetland site, an evaluated score between 0 and 3 was assigned to each of the 6 parameters. A score of 3 indicated the evaluated parameter met all the criteria required for a wetland site to be classified as 100% functional, and a score of 0 represented no functionality. Certain parameters were accordingly scored in half (0.5) increments; half increments were utilized on the point scale from 0.5 through 2.5. This allowed assessing the value of the wetland site more accurately. After assigning a value to each of the 6 parameters, the sum of all scores were then divided by 18 (the maximum combined score) to obtain a final rating, which was expressed numerically by a number between 0 and 1. A final rating score of 1 indicated that a wetland site was functioning at 100% of its capacity; whereas a final score of 0 indicated a wetland site had no functional value. WRAP scores (0–1) were used to establish disturbance gradients: high quality, (HQ, 0.75–1.0), medium quality, (MQ, 0.50–0.74) and low quality (LQ, 0–0.49). A total of 35 sites were identified in Bigodi, 21 in Kahunge, and 7 in Businge, reflecting their relative sizes and habitat heterogeneity. At each site, human activities such as agriculture, grazing, tree planting, fuelwood extraction, brick-making and domestic water use were recorded to validate disturbance gradients. Three sampling points per site were purposively selected to capture habitat variability and dominant disturbances. GPS coordinates were taken at each point, and all observed stressors were documented to define site conditions along the disturbance gradient (Table 1), (Plate 1).

Wetland-use and wetland-cover change determination

The study involved use of remote sensing. The study acquired and utilized a series of Sentinel-2 satellite imagery of 2016 and 2025 multi-temporal (cloud cover 0%–5%) sentinel images (10 m resolution). The images were downloaded from the Sentinel website (https://sentinels.copernicus.eu) on 05 July, 2025 to characterize wetland use and wetland cover changes of the wetlands under study. For each year, two

Table 1. Number of sampling sites by disturbance gradient for each wetland system based on WRAP scores.

Wetland system	WRAP score range	Disturbance gradient (WRAP score range)	Number of sampling sites
Bigodi (BiWS)	0.75–1.0	High quality (HQ)	12
	0.5–0.74	Medium quality (MQ)	20
	0.0–0.49	Low quality (LQ)	3
Kahunge (KaWS)	0.75–1.0	High quality (HQ)	3
	0.5–0.74	Medium quality (MQ)	16
	0.0–0.49	Low quality (LQ)	2
Businge (BuWS)	0.75–1.0	High quality (HQ)	1
	0.5–0.74	Medium quality (MQ)	4
	0.0–0.49	Low quality (LQ)	2



Plate 1. Characteristics of some of the assessed and scored wetland sites. Details of each plate are provided in the respective captions. Photos by Rauben and Francis.

Sentinel-2 images were downloaded with cloud cover between 0-5%, uploaded into QGIS, put to natural color (Red–Green–Blue) band order (4–3–2) and visually observed for clarity. The clearer image was used in the image classification. For 2016, the clearer adopted image was the one captured on 14th March, while for 2025, the image taken on 25th February was adopted. The adopted Sentinel satellite images were clipped to retain only the study area (area around KNP, Kamwenge district) containing the wetlands under study and classified using supervised classification procedures (maximum likelihood) by Semi-Automatic Classification plugin in QGIS 3.24.5-1 software. This was done by creating a virtual band set layer in a red–green–blue (4–3–2) band order using SCP, followed by creating regions of interest for each identifiable class in the imagery, and assigning the regions of interest to appropriate macro-classes (Classification was done using macro-classes, MCID). Six informational classes were clearly observable in the images and thus classified. This determined the magnitude of different Wetland-use and wetland-cover classes/types for 2016 and 2025. These included wetland, small-scale agriculture, built-up, woodlot, grasses/grasslands and bare surface. The description of these classes is provided in Table 2 below (Kuule et al. 2022).

Ground truthing was done using a base map loaded as a layer in QGIS to visually observe the wetland uses at various wetland sites relating it to the classified images. Field observations made during field data collection about the human activities and wetland cover at various sites was also used to confirm the truthiness of the classified image. The wetland systems were then clipped out of a larger classified image/raster by 'clip raster by mask layer' algorithm using wetland vectors obtained from 'Ug_wetlands 1996' shapefile. Results of classification accuracy algorithm indicated overall accuracy of 91.7878% (k = 0.815) and 91.8104% (k = 0.815) for 2016 and 2025, respectively (Appendices 4 and 5). Classification reports containing percentage cover and area for each WUWC class/type were obtained using the SCP classification report algorithm. This revealed use and cover changes, disturbances, or developments occurring in wetlands under study.

Spatial-temporal distribution of macrozoobenthos as indicators of habitat quality

Macrozoobenthos were sampled following a sampling protocol adopted from the Biological Monitoring Working Party (BMWP) Scoring System, modified by Paisley et al. (2004). All characteristic habitats and substrates typical of the whole wetland areas were sampled including vegetation, sand-mud (SM) and stones (for the case of the streams passing through the wetlands) biotopes. A scoop net (diameter: 30 cm, mesh size: 1 mm) attached to a long handle was placed on the water floor. Samples were taken by disturbing the substrate in front of the net using the feet and moving backwards, for a 5-minute period per sample (Correa-Araneda et al. 2014), and covering the largest number of habitats possible (water column, substrate, plant roots, etc.). Samples were collected in three main biotopes, namely vegetation, sand-mud (SM) and stones (for the case of the streams passing through the wetlands). Sampling procedures for each biotope are described below.

Vegetation: The vegetation biotope included aquatic vegetation. Aquatic vegetation referred to both emergent and submerged vegetation in the wetland, including roots and stems of floating vegetation. A net was repeatedly pushed through the submerged or floating aquatic vegetation over an area of approximately one square meter.

Sand-mud (SM): At sampling sites with SM, samples were collected from sand (0.06–2 mm) and mud, silt or clay (<0.06 mm) in available water for a period of one minute. SM biotopes were stirred by shuffling one's feet while the shuffled area was continuously swept over by a net to catch dislodged organisms.

Table 2. Description of wetland-use and wetland-cover types (classes) mapped in the study wetlands.

Class	Description
Wetland	These are areas that are permanently or seasonally waterlogged comprising of both woody and herbaceous vegetation
Small-scale agriculture	This consists of land cultivated for both subsistence and commercial purposes. It includes those areas recently sowed with crops and non-timber plantations such as coffee and sugarcane
Built-up	These are all permanent and semi-permanent buildings housing people and other infrastructure such as roads, factories and buildings in trading centers
Grasses/grasslands	These are areas dominated by tall and short grasses that are perennial. Some of the grasslands are characterized by woody vegetation and shrubs that cover less than 10% of the land area
Bare-surface	These are areas with exposed soil surfaces and no vegetation such as previously burnt areas, rock surfaces etc.

Collection of sand and mud into the net was avoided by giving few seconds for large sediment particles to settle before sweeping the net.

Stones (for the case of streams passing through the wetlands): The scoop net was placed close to and downstream of the kicked stones to allow dislodged organisms to be carried into the net by the current. Where necessary, stones were turned over to dislodge organisms. Kicking in loose stones was done for two minutes while kicking in embedded stones, in particular bedrock, was prolonged for up to a maximum of five minutes and sampling time was noted.

Hand-picking and visual observation: Substrate was visually searched for attached organisms, especially on the decaying wood and any organism seen was picked up. The macrozoobenthos collected from the different biotopes at each site were combined and transferred into a 500 ml sampling bottle containing 80% ethanol using feather-weight forceps for preservation and identified later in the laboratory.

Laboratory procedures

Sorting: Large samples were subdivided into sub-samples of nearly equal size for comfortable sorting. The sub-samples were placed in different containers with 80% ethanol and labeled. A small quantity of unsorted material was placed on a tray for an initial general sorting for larger organisms with the help of a magnifying lens. Shell fragments, vegetal debris or coarse detritus in the sample was rinsed in a separate container and checked for the presence of macrozoobenthos. Large organisms were placed immediately in appropriate containers making sure that no other smaller animals are attached to their bodies. Fine sorting was performed under a dissection microscope in laboratory at either Mountains of the Moon University or Mbarara University of Science and Technology. During this phase, a small quantity of the sample was spread onto a Petri dish and carefully examined to identify the organisms. Organisms were picked up and placed in different containers according to the main taxonomic groups. Containers were labeled inside and outside.

Identification procedures: Macroinvertebrates were identified to family level. Identification keys were used that included those developed for the East and southern African sub-region . Number of Taxa and Average score per taxon (ASPT) were determined using the Biological Monitoring Working Party (BMWP) scores. The BMWP scores were obtained by adding up the sensitivity scores of the different taxa found at a site with no consideration of abundance. Sensitivity scores were obtained from the score sheet developed by Paisley et al. (2004). TARISS scores were used for the taxa that lacked a BMWP score; however, the TARISS scores were converted to BMWP scores by the formula;

$$\text{BMWP taxa score} = \frac{TARISS\ taxa\ score}{TARISS\ max\ .\ score} \times BMWP\ max\ .\ score$$

ASPT was obtained by dividing the BMWP scores by the number of taxa at a given site i.e;

$$\text{ASPT} = \frac{Sum\ of\ BMWP\ scores\ at\ a\ site}{Number\ of\ taxa\ at\ that\ site}$$

Physico-chemical parameters of the wetland waters along a disturbance gradient

Water samples for laboratory analysis were collected in 500 ml plastic bottles previously disinfected, rinsed with distilled water and dried. A 500 ml plastic bottle was immersed to a maximum possible depth (5–30 cm, based on depth of water at the sampling site) and was covered while still in water. At each sampling site, water samples were collected from three collection points to form a composite (Dorche et al. 2019). During transport and until commencement of the investigations, the water samples were stored in cool conditions at around 4 °C (De Troyer et al. 2016) within an ice box. Some physico-chemical parameters were tested in situ, and these included pH, Temperature, Dissolved Oxygen (DO), total dissolved solids, salinity, electrical conductivity (EC) and Turbidity using a well-calibrated HANNA multi-parameter instrument. The chemical ionic species; nitrates, nitrites, phosphates, ammonium, as well as hardness, were analytically determined in a laboratory as fast as possible within 48 hours after sample collection to avoid contamination. The analysis was performed as follows (Palintest Ltd, 2018).

Nitrates NO_{3^-} (mg/l): Nitrate in water was first reduced to nitrite, using zinc-based nitrate powder in combination with Nitratest tablet, which aids rapid flocculation for a one-minute contact period. The resulting nitrite from the reduction stage was then determined in a diazonium reaction by reaction with sulphanilic acid in the presence of *N*-(1-naphthyl)-ethylenediamine (second Nitratest tablet). The procedure was as follows: a 20 ml filtered water sample was placed in a tube, a spatula end-full of zinc-based nitrate powder was added, one nitratest tablet was added and the contents were mixed vigorously, allowed to settle for 5 minutes to clear up. 10 ml of the settled clear sample was decanted into another tube where the second Nitratest tablet was added, mixed and allowed to settle for color development. Nitrate was then measured in mg/l using a Palintest photometer.

Nitrite (mg/l): The principle for nitrite test followed an adaptation of the Ferrous Sulfate method in which the reaction between nitrite and the reagent causes a color change in the sample. The intensity of color was determined by a compatible photometer and the concentration was displayed in mg/L of nitrite.

Phosphate (mg/l): The phosphate in the water sample was reacted under acid conditions with ammonium molybdate to form phospho-molybdic acid. This compound was reduced by ascorbic acid in the presence of a catalyst to form the intensely colored 'molybdenum blue' complex. The procedure was done by filtering the water sample through a Whatman filter paper. 10 ml of the filtrate was reacted with phosphate LR reagent, number 1 tablet, which was mixed to dissolve completely, followed by phosphate LR reagent, number 2 tablet and mixed to dissolve. The sample was allowed to settle for 10 minutes for full color development. The intensity of the color produced in the test is proportional to the phosphate concentration, and was measured using a Palintest Photometer, recorded in mg/l.

Ammonium (mg/l): The reagents used follow the Nessler Method, in which the reaction between ammonia and reagents cause a yellow tint in the sample. The intensity of color was determined by a compatible photometer and the concentration was displayed. The results were displayed in (mg/l) of ammonium.

Total hardness (mg/l): The test used was a colorimetric method, where under the controlled conditions calcium and magnesium ions react in water with Hardicol indicator mixed to produce a purple coloration. The procedure was done as follows; 10 ml filtered water sample was mixed with hardness tablet reagent (Table 1), mixed well to fully dissolve, then followed by hardness tablet 2, mixed well to dissolve too. The solution was left to stand for two minutes to allow full color development. The intensity of the color is proportional to the total hardness of the water and was measured using a Palintest Photometer. The Total Hardness result is displayed in mg/l.

Data analysis

Wetland-use and wetland-cover change analysis

Analysis of the collected data was comprehensively done to interpret trends, quantify changes and understand the dynamics of wetland-cover over time for the wetlands under study around the park. Geographical Information System (GIS) software for spatial analysis and visualization of results was utilized. Sentinel-2 images were imported into QGIS as Raster data. Appropriate sentinel band analysis was selected and virtual raster built and WUWC types classified. The classified image was saved by exporting as an image. Statistical data in the form of percentage cover from classification reports were studied to identify which WUWC type expanded or contracted over time (2016 to 2025).

Spatial-temporal distribution of macrozoobenthos as indicators of habitat quality

Macrozoobenthos taxa abundance data were entered in MS Excel ver. 2019 and used to compute the relative abundances of the different taxa. Macrozoobenthos classified to family level were described by relative abundance and Shannon's diversity index (H′). Shannon's diversity index was computed using PAST ver. 4.11 computer software. Families' diversity values for all sites across disturbance gradients and seasons were tested for normality using Anderson-Darling normality test in Minitab software ver. 20. Site differences in family diversity were tested using Kruskal–Wallis test for normally distributed data and One-way ANOVA test for not normally distributed data from Minitab ver. 20 statistical software. Pairwise comparison test for Kruskal-Wallis was carried out to identify where significant differences particularly existed. Seasonal variation in macrozoobenthos family diversity and ASPT were analyzed using paired

samples t-test (for normally distributed data) and Wilcoxon ranked test (for data not normally distributed) in Minitab ver.20 software and lastly the family diversity (H′) was related to WRAP scores per site using Spearman's rank correlation. All analyzes were performed at a 5% level of significance.

Physico-chemical parameters of the wetland waters along a disturbance gradient

Data of physico-chemical parameters were entered into Minitab ver. 20 software. The data were tested for normality across disturbance gradients and sampling seasons using the Anderson-Darling normality test. Mean, standard deviation and median for each physico-chemical parameter was computed. The mean or median concentrations of parameters that did not fall within NEMA standards were compared with their standards as prescribed by NEMA using a one-sample t-test or sign test. The comparison was made because NEMA standards provide the relevant national regulatory reference framework for evaluating whether measured water-quality conditions fall within acceptable limits for environmental protection and aquatic ecosystem management in Uganda. The standards were therefore used as a regulatory benchmark, rather than as a substitute for ecological assessment. A principal component analysis was used to cluster study sites based on physical chemical parameters values across seasons for each wetland system studied and lastly, WRAP scores were correlated with nutrient content (nitrates) in the wetlands using Spearman's rank correlation. All the above tests were done in Minitab ver.20 software at a 5% level of significance.

Results

Wetland-use and wetland-cover change (2016–2025)

Six Wetland-use and wetland-cover (WUWC) classes were identified across the studied wetland systems: wetland vegetation, small-scale agriculture, grassland, built-up areas, woodlots and bare surfaces. Spatial analysis of Sentinel-2 imagery revealed varying degrees of change among the three wetland systems, Bigodi, Kahunge and Businge, between 2016 and 2025. Overall, the magnitude and direction of WUWC change differed markedly among the wetlands, reflecting contrasting levels of landscape transformation, anthropogenic influence and wetland management.

Bigodi wetland system showed minimal Wetland-use and wetland-cover change over the study period. Wetland vegetation remained dominant, increasing slightly from 58.72% in 2016 to 59.89% in 2025 (+1.17%). Small-scale agriculture declined marginally from 16.40% to 16.19%, while bare surfaces also decreased slightly. Built-up areas remained negligible (<1%) (Figures 2 and 3). Overall, Wetland-use and wetland-cover change patterns in Bigodi remained largely stable throughout the study period.

Within Bigodi Wetland Sanctuary, a part of Bigodi wetland system that is protected by a community association, Kibale Association for Rural and Environmental Development (KAFRED), wetland cover remained high (>77%) with only a slight decrease (–1.54%) and a notable increase in woodlot cover (+3.17%) while bare surfaces declined (Figures 4 and 5). Despite minor changes, wetland vegetation continued to highly dominate the sanctuary landscape.

Kahunge wetland system experienced moderate alteration, with wetland cover decreasing by 3.58% and small-scale agriculture reducing by 3.5%. Grassland cover increased by 5.23%, indicating a shift from wetland and previous agricultural areas to grasses either through natural change or human management. Built-up areas increased slightly but remained minimal, suggesting low-level development (Figures 6 and 7).

Businge wetland system showed the greatest transformation. Wetland cover sharply declined (–8.47%), while grasslands expanded (+7.64%), replacing lost wetland and agricultural areas. Agriculture remained dominant but declined slightly (–4.8%). Built-up areas increased marginally. Overall, Businge exhibited the highest wetland loss and ecosystem conversion. (Figures 8 and 9).

Spatial-temporal distribution of macrozoobenthos as indicators of habitat quality

Taxonomic composition and abundance

A total of 1943 macrozoobenthic individuals were recorded across all wetlands. Of these: Bigodi had 1189 individuals (22 families), Kahunge had 538 individuals (21 families) and Businge had 216 individuals (21 families).

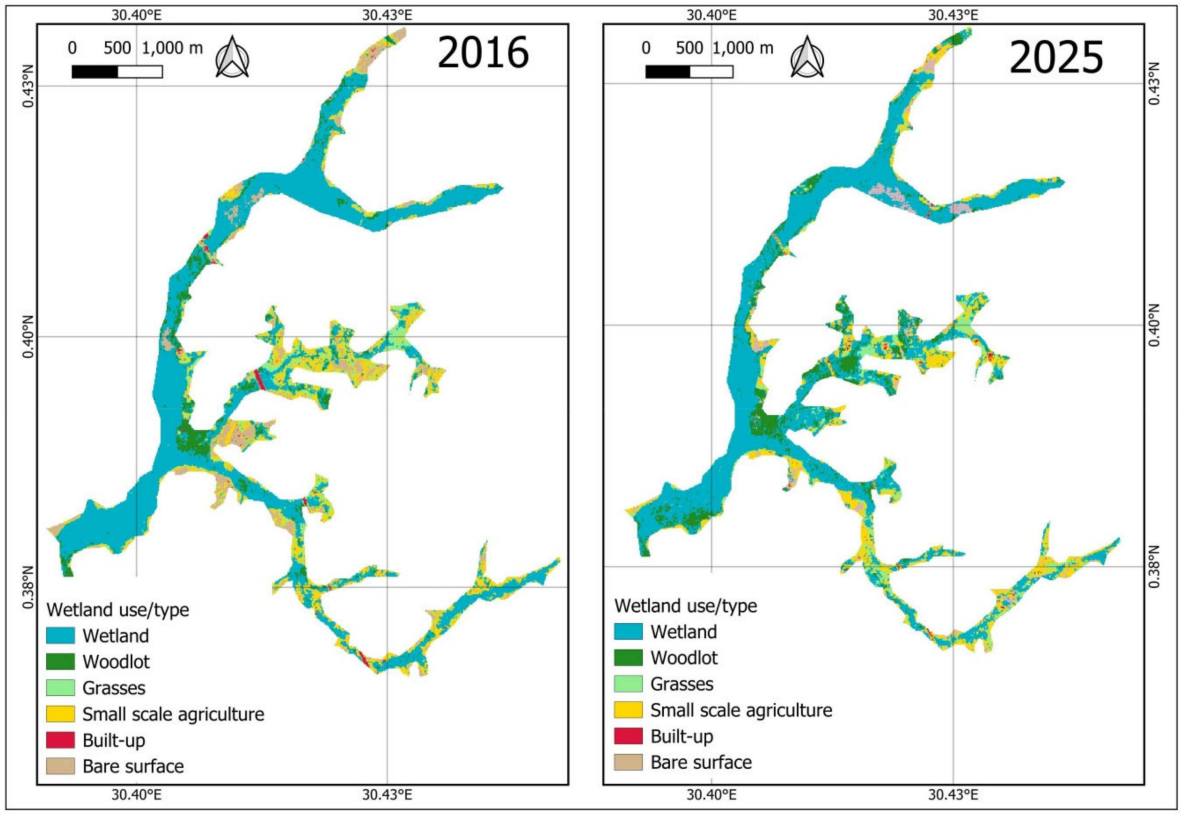


Figure 2. The main wetland-use and wetland-cover types in Bigodi wetland system between 2016 and 2025.

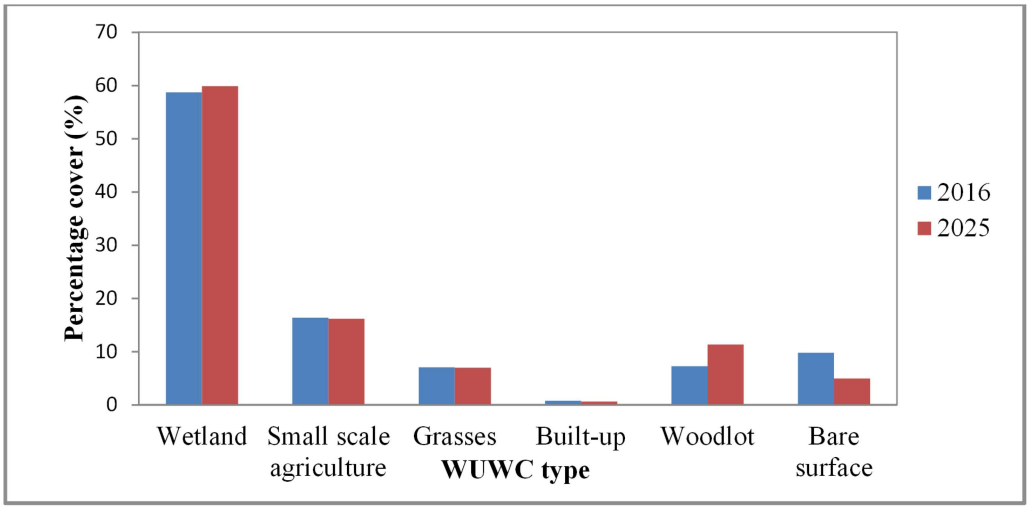


Figure 3. Wetland-use and wetland-cover types and their percentage cover in Bigodi wetland system for the year 2016 and 2025.

Bigodi contributed 61.2% of total individuals, Kahunge 27.7% and Businge 11.1%. Hence, Bigodi had the highest abundance and taxonomic richness.

In Bigodi, dominant families included Planorbidae (19.26%) and Ampullaridae (17.41%). In Kahunge, Coenagrionidae (15.61%) and Ampullaridae (10.22%) were most abundant. In Businge, Coenagrionidae dominated strongly (25%), followed by Gomphidae and Naucoridae (11.11% each) (Table 3).

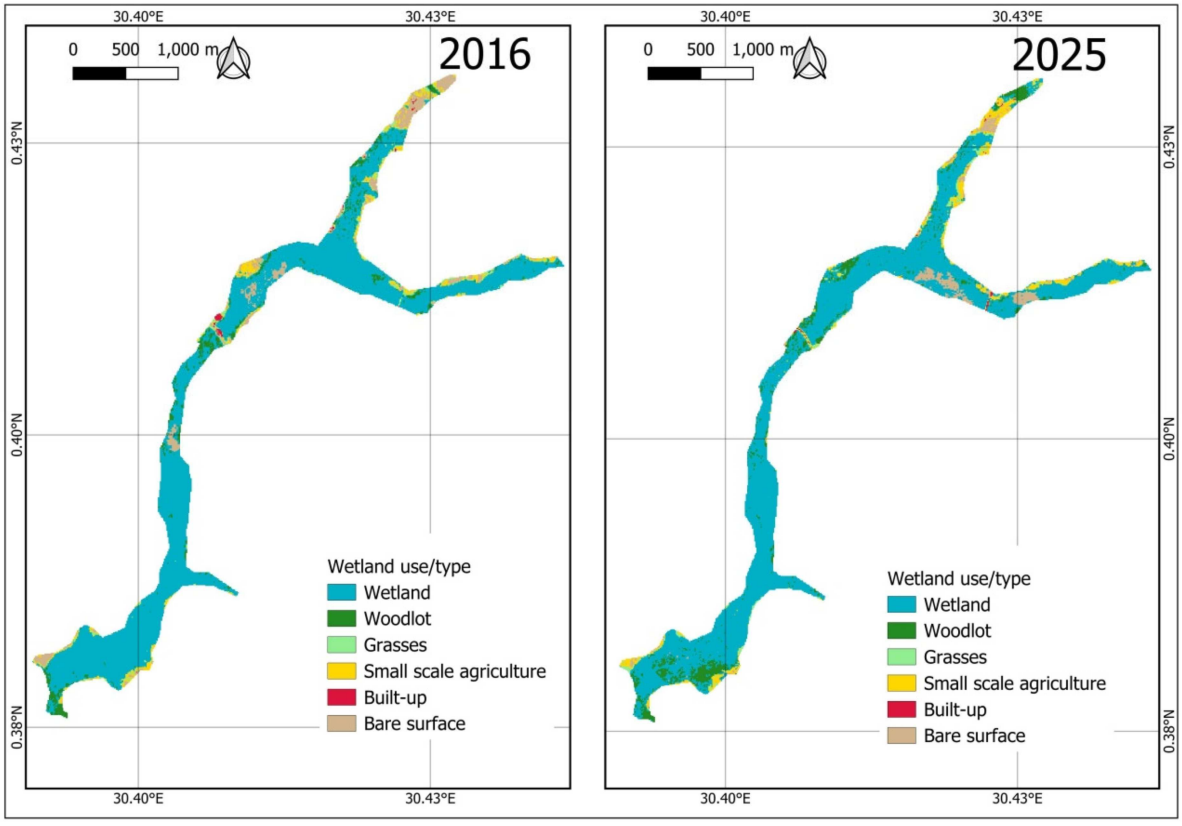


Figure 4. The main Wetland-use and wetland-cover types in Bigodi wetland sanctuary between 2016 and 2025.

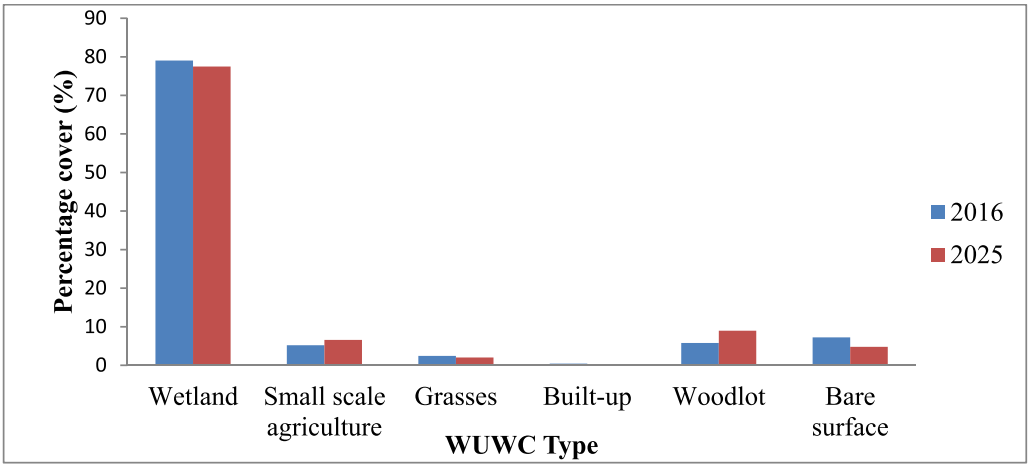


Figure 5. Wetland-use and wetland-cover types and their percentage cover in Bigodi wetland sanctuary for the year 2016 and 2025.

Family diversity (H′) across disturbance gradients and seasons

Shannon diversity (H′) varied across wetlands and disturbance gradients.

In Bigodi, there were no statistically significant differences in family diversity across high-quality (HQ), moderate-quality (MQ) and low-quality (LQ) sites (One-way ANOVA; $p = 0.257$; $F = 1.39$; $DF = 2$). Diversity remained relatively stable across gradients. However, HQ sites had the highest mean family diversity ($H' = 1.3140 \pm 0.3384$) while LQ sites had the lowest ($H' = 1.051 \pm 0.661$). There was a relatively higher median family diversity in the dry season ($H' = 1.3340$) than the wet season ($H' = 1.3090$) but

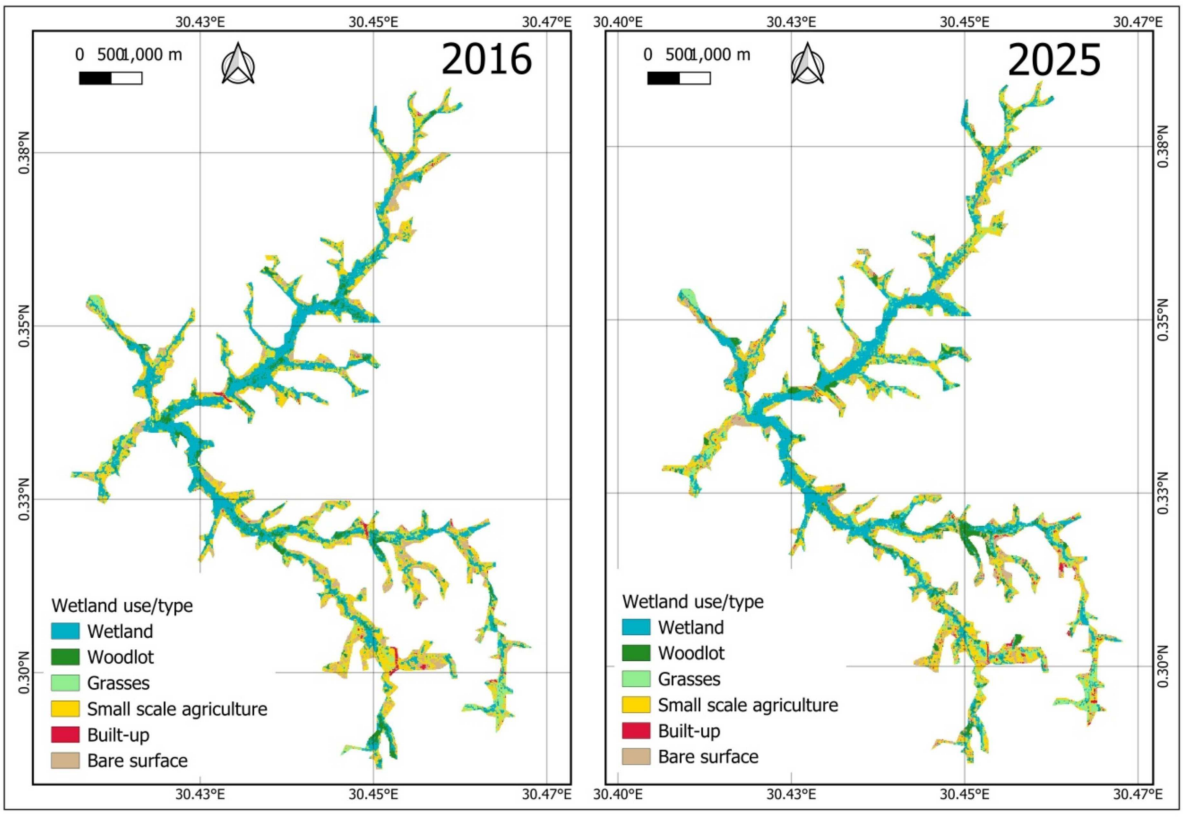


Figure 6. The main Wetland-use and wetland-cover types in Kahunge wetland system between 2016 and 2025.

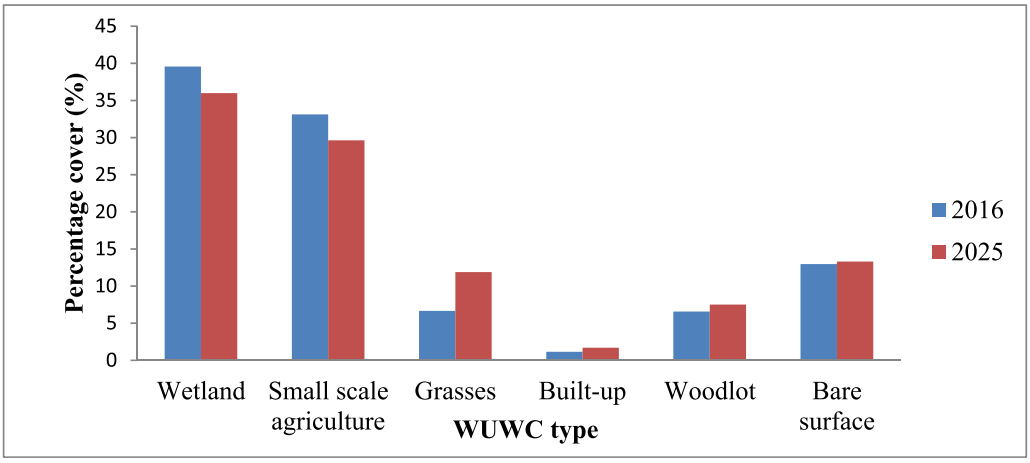


Figure 7. Wetland-use and wetland-cover types and their percentage cover in Kahunge wetland system for the year 2016 and 2025.

analysis with the Wilcoxon signed-rank test showed no significant difference in family diversity between the dry and wet seasons ($p = 0.644$).

In Kahunge, seasonal variation in diversity was significant ($p = 0.002$) and disturbance gradient differences were not statistically significant. Wilkoxon signed rank test showed a significantly higher family diversity in the dry season than wet season ($p = 0.002$), while Kruskal–Wallis test indicated no significant difference in family diversity across disturbance gradients ($p = 0.545$; $H = 1.21$; $DF = 2$), with LQ sites showing lowest median family diversity ($H' = 0.9906$).

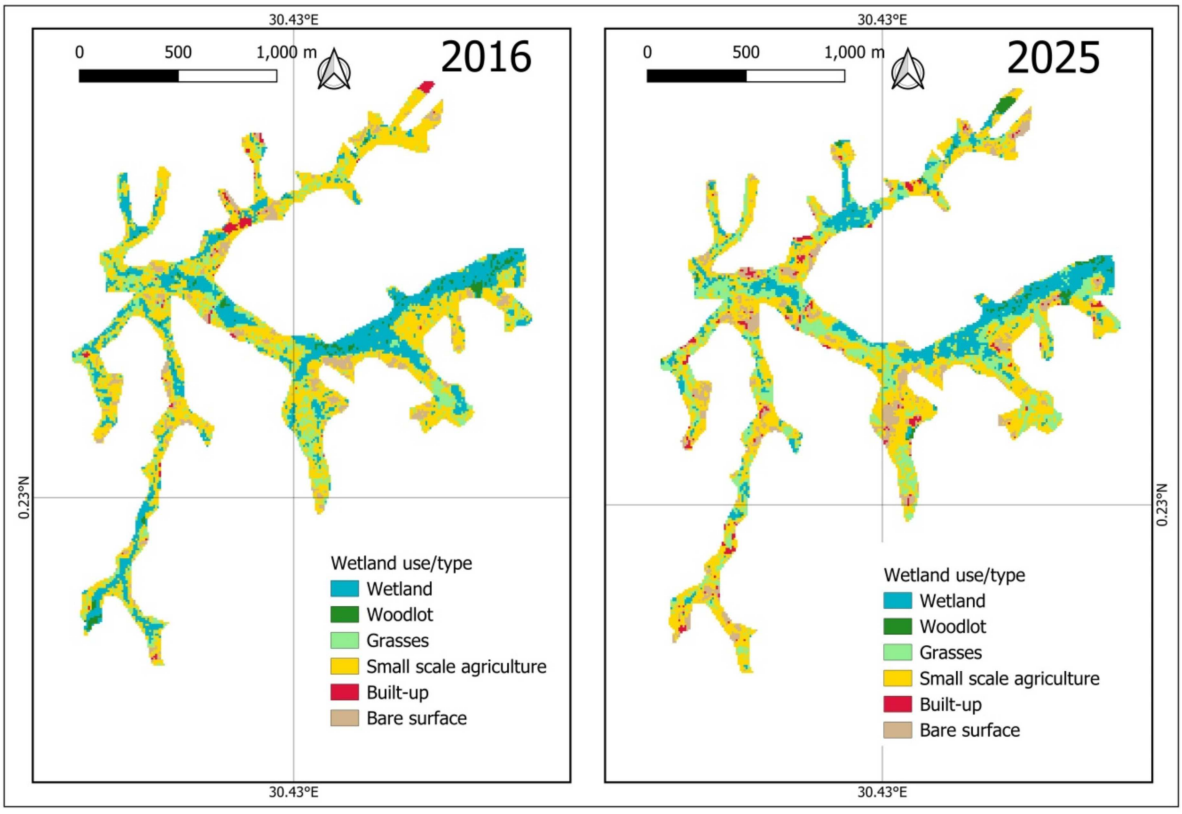


Figure 8. The main Wetland-use and wetland-cover types in Businge wetland system between 2016 and 2025.

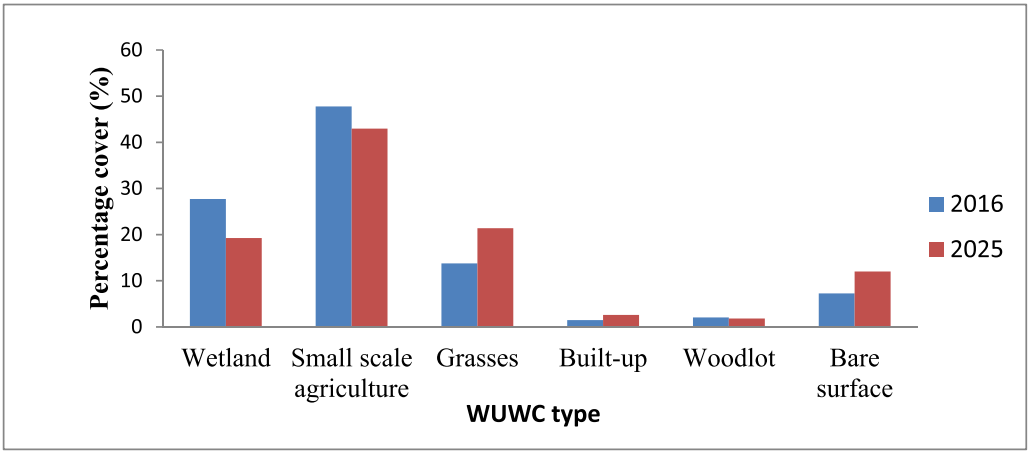


Figure 9. Wetland-use and wetland-cover types and their percentage cover in Businge wetland system for the year 2016 and 2025.

In Businge, seasonal differences in diversity were significant ($p = 0.033$). Paired samples t-test showed that family diversity was significantly higher in the dry season (1.4209 ± 0.3088) than in the wet season (1.1753 ± 0.2883). Diversity values were lower at LQ sites compared to HQ sites (Figure 10).

Average score per taxon (ASPT)

In Bigodi and Kahunge wetland systems, there was no statistically significant ASPT difference across disturbance gradients ($p > 0.05$).

Table 3. The abundances and relative abundances of orders/class and families of macrozoobenthos sampled from the wetland systems: Bigodi, Kahunge and Businge in the months of January and April 2025.

		Abundance per wetland system			Relative abundance (%)		
Orders/Class	Family/subclass	Bigodi	Kahunge	Businge	Bigodi	Kahunge	Businge
Coleoptera	Dytiscidae	20	32	8	1.68	5.95	3.7
	Elmidae	1	10	8	0.08	1.86	3.7
	Hydrophillidae	46	35	4	3.87	6.51	1.85
	Gyrinidae	-	-	2	-	-	0.93
	Limnichidae	4	-	-	0.34	-	-
	Scirtidae	13	29	-	1.09	5.39	-
Crustacea	Atyidae	87	8	1	7.32	1.49	0.46
Diptera	Chironomidae	82	20	14	6.9	3.72	6.48
	Syrphidae	15	28	9	1.26	5.2	4.17
	Tipulidae	-	3	1	-	0.56	0.46
	Culicidae	-	-	2	-	-	0.93
Ephemeroptera	Baetidae	4	-	12	0.34	-	5.56
Hemiptera	Belastomatidae	35	5	4	2.94	0.93	1.85
	Naucoridae	41	53	24	3.45	9.85	11.11
	Nepidae	34	20	4	2.86	3.72	1.85
Molluca	Ampullariidae	207	55	3	17.41	10.22	1.39
	Bulinidae	40	2	1	3.36	0.37	0.46
	Lymnaeidae	77	33	10	6.48	6.13	4.63
	Planorbidae	229	39	20	19.26	7.25	9.26
	Sphaeriidae	35	9	-	2.94	1.67	-
Odonata	Aeshnidae	8	9	-	0.67	1.67	-
	Coenagrionidae	95	84	54	7.99	15.61	25
	Gomphidae	85	39	24	7.15	7.25	11.11
	Libellulidae	26	24	10	2.19	4.46	4.63
Trichoptera	Philopotamidae	5	1	-	0.42	0.19	-
Plecoptera	Leuctridae	-	-	1	-	-	0.46

Notes: (-) means taxa not found in the wetland system.

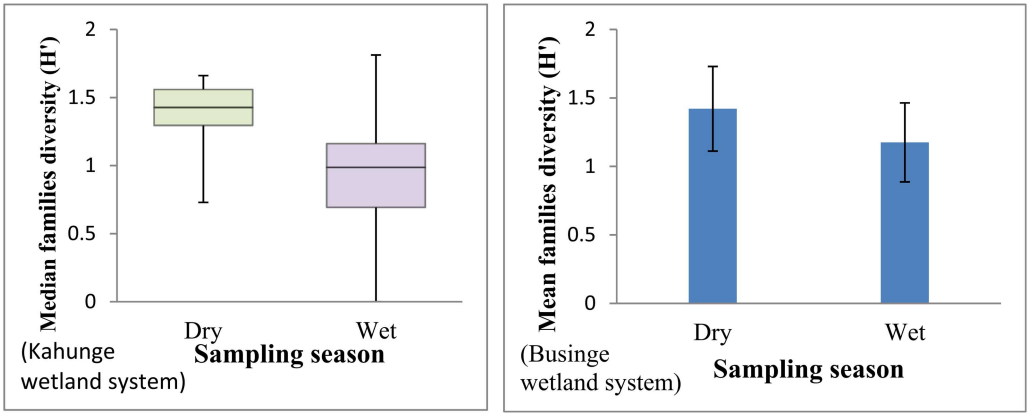


Figure 10. Median family diversity of Kahunge wetland system and mean family diversity of Businge wetland system across sampling seasons.

In Businge, ASPT differed significantly across disturbance gradients (Kruskal–Wallis; $p = 0.004$, H = 10.92, DF = 2). Paired samples t-test showed a significantly higher ASPT in wet season (4.851 ± 1.114) than dry season (4.238 ± 0.583) ($p = 0.048$, T = −2.25). Lower ASPT values were recorded in low-quality sites. The pairwise comparison in Kruskal–Wallis indicated that the median ASPT of HQ was significantly higher than that of MQ and LQ while median ASPTs of MQ and LQ were not significantly different (Table 4) (Figure 11).

Relationship between WRAP scores and diversity

Spearman's rank correlation indicated positive associations between WRAP scores and family diversity. The relationship was strongest in Businge, where higher WRAP scores corresponded with higher diversity values. In Bigodi, the relationship was weaker but positive (Figure 12).

Table 4. Median family diversity and ASPT across disturbance gradients in Businge wetland system.

Variable	Disturbance gradient	Minimum	Q1	Median	Q3	Maximum
ASPT	HQ	5.029	5.029	5.652[a]	6.275	6.276
	LQ	3.433	3.555	3.94[b]	4.19	4.267
	MQ	3.422	3.632	4.417[b]	4.967	5.55

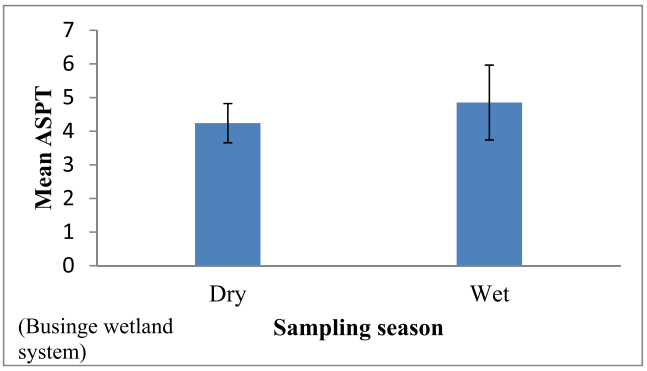


Figure 11. Mean ASPT across sampling seasons in Businge wetland system.

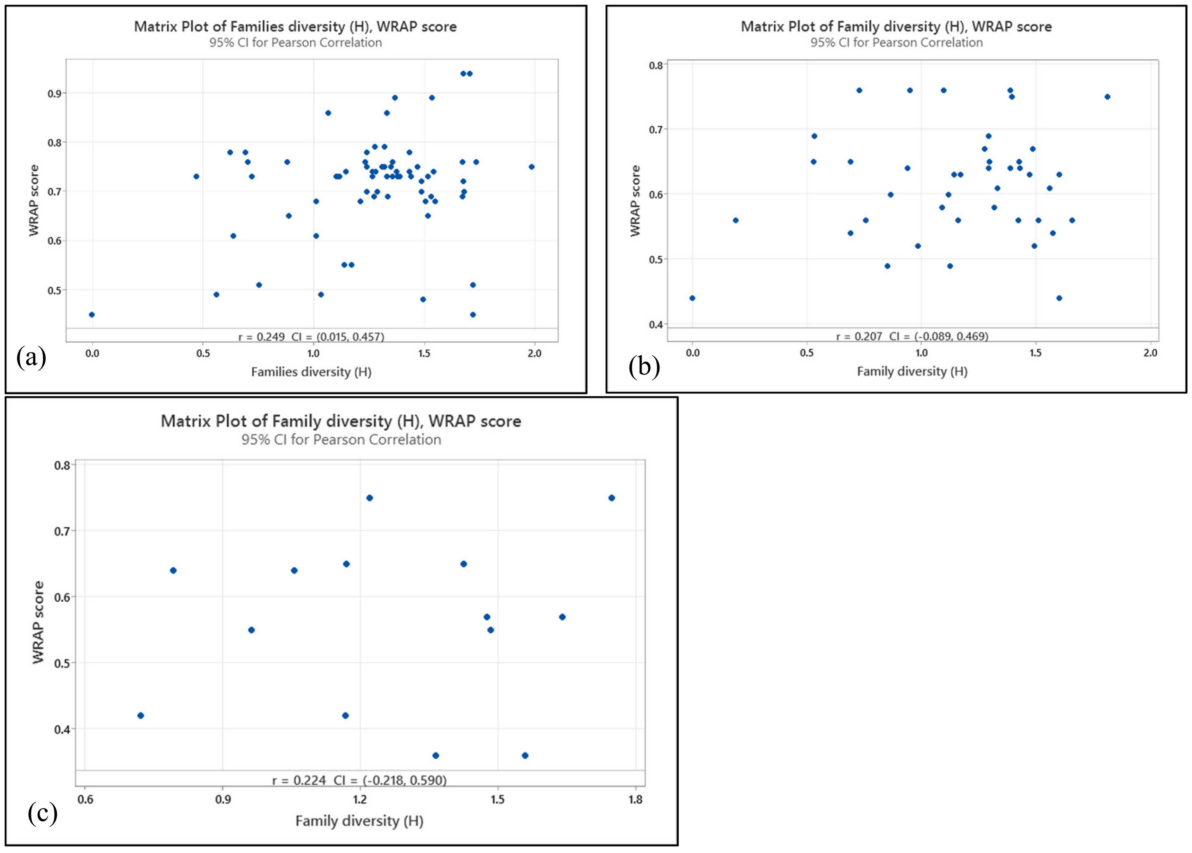


Figure 12. Pearson correlation plots between WRAP scores and family diversity for Bigodi wetland system (a), Kahunge wetland system (b) and Businge wetland system (c).

Physico-chemical parameters

General water quality status

Across wetlands, mean values of pH, dissolved oxygen, TDS, electrical conductivity, turbidity, salinity, hardness and nutrients were generally within National Environment Management Authority (NEMA) standards. Isolated exceedance was observed for nitrites in certain low-quality sites.

Generally, across sampling seasons and the subsequent HQ–MQ–LQ, parameters remained within standards with no consistent exceedance pattern by gradient. Based on these variables, water quality did not show a systematic degradation trend along the disturbance gradient within systems.

Seasonal variation

Significant seasonal differences were observed in selected parameters. For instance, in dry season, higher hardness and nitrate concentrations in Businge (mean nitrate = 0.413 ± 0.393 mg/L; hardness = 175 ± 144.3 mg/L) were recorded. In the wet season, Bigodi recorded higher TDS (142.5 ± 80.6 ppm) and nitrate (0.675 ± 0.496 mg/L) as shown in Table 5 above.

Seasonal clustering was confirmed by Principal Component Analysis, which separated wetlands based on physico-chemical signatures.

Principal component analysis (PCA) of dry season physico-chemical parameters explained 49.1% of the total variance and revealed partial separation among the three wetland systems. Businge (BuWS) was clearly distinguished along PC1, whereas Bigodi (BiWS) and Kahunge (KaWS) showed some overlap but remained distinguishable. Businge was positively associated with PC1 and characterized by comparatively higher nitrate (NO_{3^-}), hardness and pH values. In contrast, Bigodi was negatively associated with PC1 and exhibited lower values of these parameters. Kahunge overlapped with Bigodi along PC1 but was characterized by relatively higher turbidity. The PCA for the wet season showed that Bigodi wetland system (red) was separated from the other two wetland systems along PC1, suggesting that it had distinct characteristics. Businge (green) and Kahunge (blue) wetland systems overlapped to some extent, indicating similarities between them. TDS and NO_{3^-} were strongly correlated with PC1, while pH, PO43- and NH4 + were more closely related to PC2 (Figure 13).

Nutrients and disturbance gradient

Spearman correlation analysis showed that nitrate concentrations were negatively associated with WRAP scores across all wetland systems. That is to say, $r = -0.064$ for BiWS, $r = -0.158$ for KaWS) and $r = -0.238$ for BuWS. Lower ecological quality sites exhibited relatively higher nutrient concentrations (Figure 14).

Discussion

Wetland-use and wetland-cover change from 2016 to 2025

Wetland-use and wetland-cover trajectories differed markedly among Bigodi, Kahunge and Businge, indicating that wetlands located within the same protected-area buffer can experience contrasting ecological pressures and management outcomes. Rather than representing a uniform process of wetland degradation, the observed changes suggest that wetland condition is shaped by the interaction of land-use pressure, local governance, accessibility, livelihood dependence and the spatial distribution of human activities. This interpretation is consistent with evidence that land-use change around protected areas is often spatially heterogeneous because surrounding communities, markets, infrastructure and conservation institutions exert different pressures across relatively short distances (Southworth et al. 2010; Davis and Hansen 2011).

In Bigodi, the relatively stable wetland cover and the slight increase in wetland vegetation suggest that large-scale conversion was limited during the study period. This pattern is ecologically important because the maintenance of wetland vegetation can help preserve hydrological regulation, sediment retention and habitat structure, even where some localized human activities continues. The observed stability should, however, not be interpreted as evidence that the wetland is free from disturbance. Rather, it indicates that the magnitude of landscape-level conversion was comparatively limited between 2016 and 2025. The presence of Kibale Association For Rural and Environmental Development (KAFRED) and the associated conservation and ecotourism activities provide a plausible institutional context for this relative stability, because local conservation organizations can increase community participation and create incentives for maintaining wetland resources (Douglas and Victor 2022). Nevertheless, because this study did not experimentally compare protected and unprotected sites before and after implementation of specific

Table 5. Means ± standard deviation of water physico-chemical parameters across sampling seasons and subsequent disturbance gradients for the three wetland systems; Bigodi, Kahunge and Businge.

			Mean ± standard deviation per disturbance gradient			
Wetland system	Physico-chemical parameter	Sampling season	LQ	MQ	HQ	NEMA Standard
Bigodi	Temperature (°C)	Dry	23.2 ± 0.458	21.789 ± 1.551	22.183 ± 1.542	20.0–35.0
		Wet	21.023 ± 0.752	20.335 ± 2.92	20.07 ± 1.776	
	pH	Dry	6.78 ± 0.1323	6.759 ± 0.2345	6.7258 ± 0.2341	6.0–8.0
		Wet	6.913 ± 0.337	6.5387 ± 0.2817	6.6117 ± 0.1786	
	Dissolved oxygen (ppm)	Dry	1.593 ± 0.247	2.003 ± 0.615	1.936 ± 0.452	
	Electrolytic conductivity (µS/cm)	Dry	141 ± 25	215.8 ± 116.6	188 ± 36	1500
		Wet	207.67 ± 14.36	284.6 ± 158.2	237.3 ± 53.4	
	Total dissolved solids, (ppm)	Dry	71 ± 12.49	111.1 ± 62.5	93.5 ± 17.68	1200
		Wet	103.67 ± 8.96	142.5 ± 80.6	116.5 ± 25.06	
	Salinity (PSU)	Dry	0.0667 ± 0.0153	0.1027 ± 0.0561	0.0875 ± 0.0155	
		Wet	0.1167 ± 0.0058	0.151 ± 0.0583	0.1733 ± 0.1075	
	Turbidity (FNU)	Dry	12.3 ± 2.35	11.36 ± 4.74	8.89 ± 4.35	300
		Wet	6.9 ± 10.83	7.64 ± 4.95	8.39 ± 5.47	
	Nitrates (mg/l)	Dry	0.2983 ± 0.059	0.1249 ± 0.0762	0.0767 ± 0.0451	10
		Wet	0.563 ± 0.386	0.675 ± 0.496	0.4828 ± 0.3356	
	Nitrites (mg/l)	Dry	10.33 ± 4.51	3.35 ± 3.117	5 ± 5.06	20
		Wet	3 ± 1	6.3 ± 8.47	4.167 ± 2.758	
	Phosphate (mg/l)	Dry	0.36 ± 0.052	1.495 ± 1.092	1.487 ± 0.841	10
		Wet	0.0667 ± 0.1069	0.281 ± 0.2952	0.646 ± 0.789	
	Ammonium (mg/l)	Dry	0.1467 ± 0.1193	0.148 ± 0.0623	0.1375 ± 0.0609	
		Wet	0.06 ± 0.03	0.1365 ± 0.0656	0.0958 ± 0.083	
	Hardness	Dry	50 ± 0	51.25 ± 38.83	44.17 ± 11.04	
		Wet	65 ± 8.66	91.15 ± 27.3	95.8 ± 42.6	
Kahunge	Temperature (°C)	Dry	22.775 ± 0.202	22.303 ± 1.307	21.85 ± 1.04	20.0–35.0
		Wet	19.96 ± 0.242	20.507 ± 2.511	19.502 ± 0.667	
	pH	Dry	6.615 ± 0.0058	6.6656 ± 0.3224	6.593 ± 0.301	6.0–8.0
		Wet	6.372 ± 0.234	6.5784 ± 0.3303	6.238 ± 0.289	
	Dissolved oxygen, DO (ppm)	Dry	0.655 ± 0.1443	1.072 ± 0.631	0.863 ± 0.526	
	Electrolytic conductivity	Dry	141 ± 91.2	146.6 ± 63.5	116 ± 58.6	1500
		Wet	129.5 ± 23.7	191.8 ± 76.3	144 ± 69.3	
	Total dissolved solids, TDS (ppm)	Dry	69.5 ± 45.6	73.88 ± 31.16	58 ± 29.3	1200
		Wet	64 ± 13.86	95.69 ± 37.14	70 ± 33	
	Salinity (PSU)	Dry	0.07 ± 0.0462	0.0713 ± 0.0303	0.0567 ± 0.0306	
		Wet	0.1033 ± 0.0153	0.1169 ± 0.0311	0.12 ± 0.0116	
	Turbidity (FNU)	Dry	11.05 ± 8.6	11.34 ± 4.89	15.72 ± 3.31	300
		Wet	9.95 ± 8.83	9.36 ± 5.03	8.63 ± 7.6	
	Nitrates (mg/l)	Dry	0.247 ± 0.037	0.2397 ± 0.2091	0.232 ± 0.176	10
		Wet	0.153 ± 0.0924	0.0722 ± 0.0396	0.0853 ± 0.0517	
	Nitrites (mg/l)	Dry	20.5 ± 13.08	9.69 ± 4.61	3 ± 1.155	20
		Wet	3.56 ± 4.55	1.667 ± 1.155	1.5 ± 0.577	
	Phosphate (mg/l)	Dry	0.697 ± 0.401	0.4294 ± 0.3311	0.26 ± 0.0924	10
		Wet	0.707 ± 0.619	0.1538 ± 0.0976	0.09 ± 0.0462	
	Ammonium (mg/l)	Dry	0.1133 ± 0.1041	0.1656 ± 0.0765	0.13 ± 0.0808	
		Wet	0.11 ± 0.0346	0.0819 ± 0.0483	0.0533 ± 0.0533	
	Hardness	Dry	117.5 ± 77.9	61.9 ± 73.4	61.7 ± 20.2	
		Wet	130 ± 11.55	85.94 ± 36.48	78.33 ± 7.64	
	Temperature (°C)	Dry	22.018 ± 0.615	22.125 ± 0.861	22.45 ± 0.825	20.0–35.0
Businge		Wet	20.24 ± 0.531	21.948 ± 1.944	20.38 ± 1.5	
	pH	Dry	7.745 ± 0.0866	7.592 ± 0.857	7.6 ± 0.06	6.0–8.0
		Wet	7.155 ± 0.014	7.1925 ± 0.097	7.125 ± 0.042	
	Dissolved oxygen, DO (ppm)	Dry	0.28 ± 0.1501	0.4 ± 0.32	0.56 ± 0.5	
	Electrolytic conductivity	Dry	223 ± 12.7	342.5 ± 50.3	287 ± 24.1	1500
		Wet	235.5 ± 8.66	205 ± 22	182 ± 27	
Businge	Total dissolved solids, TDS (ppm)	Dry	112 ± 6.93	170 ± 25.4	144 ± 12.4	1200
		Wet	115.5 ± 2.89	101.75 ± 10.69	91 ± 5.43	
	Salinity (PSU)	Dry	0.105 ± 0.0058	0.1675 ± 0.0263	0.14 ± 0.0132	
		Wet	0.125 ± 0.0058	0.115 ± 0.0192	0.12 ± 0.002	
	Turbidity (FNU)	Dry	11.95 ± 1.79	6.9 ± 5.34	14.3 ± 2.86	300
		Wet	8.1 ± 4.16	10.65 ± 6.45	10.3 ± 3.25	
	Nitrates (mg/l)	Dry	0.413 ± 0.393	0.3232 ± 0.1702	0.17 ± 0.0186	10
		Wet	0.086 ± 0.0162	0.0925 ± 0.0921	0.014 ± 0.0174	
	Nitrites (mg/l)	Dry	20.6 ± 14.43	8 ± 2.45	6.3 ± 1.36	20
		Wet	2.5 ± 1.732	3.75 ± 0.957	0 ± 0	
	Phosphate (mg/l)	Dry	1.5 ± 1.396	1.15 ± 0.875	0.37 ± 0.427	10
		Wet	0.2575 ± 0.0954	0.23 ± 0.052	0.175 ± 0.0404	
	Ammonium (mg/l)	Dry	0.22 ± 0.0346	0.1275 ± 0.0499	0.18 ± 0.0193	
		Wet	0.18 ± 0.0981	0.125 ± 0.0311	0.045 ± 0.2321	
	Hardness	Dry	175 ± 144.3	138.8 ± 63.4	80 ± 17.8	
		Wet	107.5 ± 31.8	108.8 ± 59.2	105 ± 17.5	

Notes: Highlighted in red indicate means that fall out of NEMA standards.

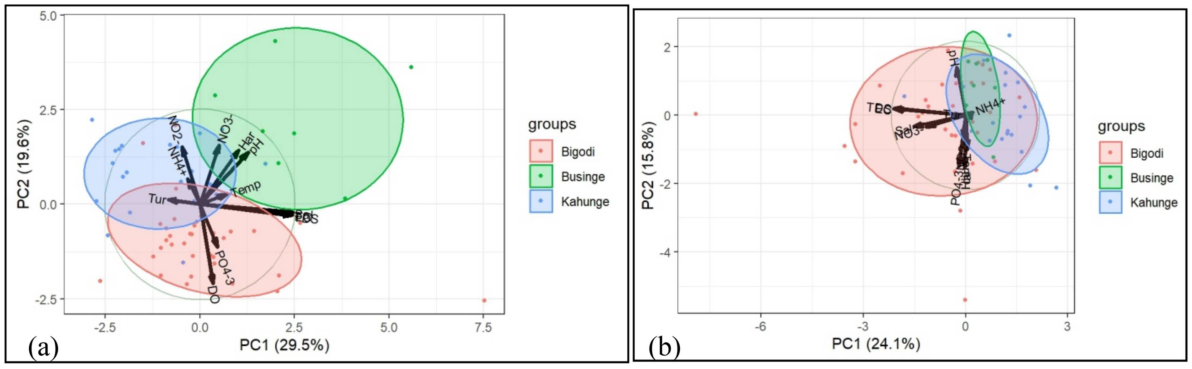


Figure 13. Principle component analysis output of the three wetland systems based on values of physico-chemical parameters obtained in the dry season (a) and wet season (b).

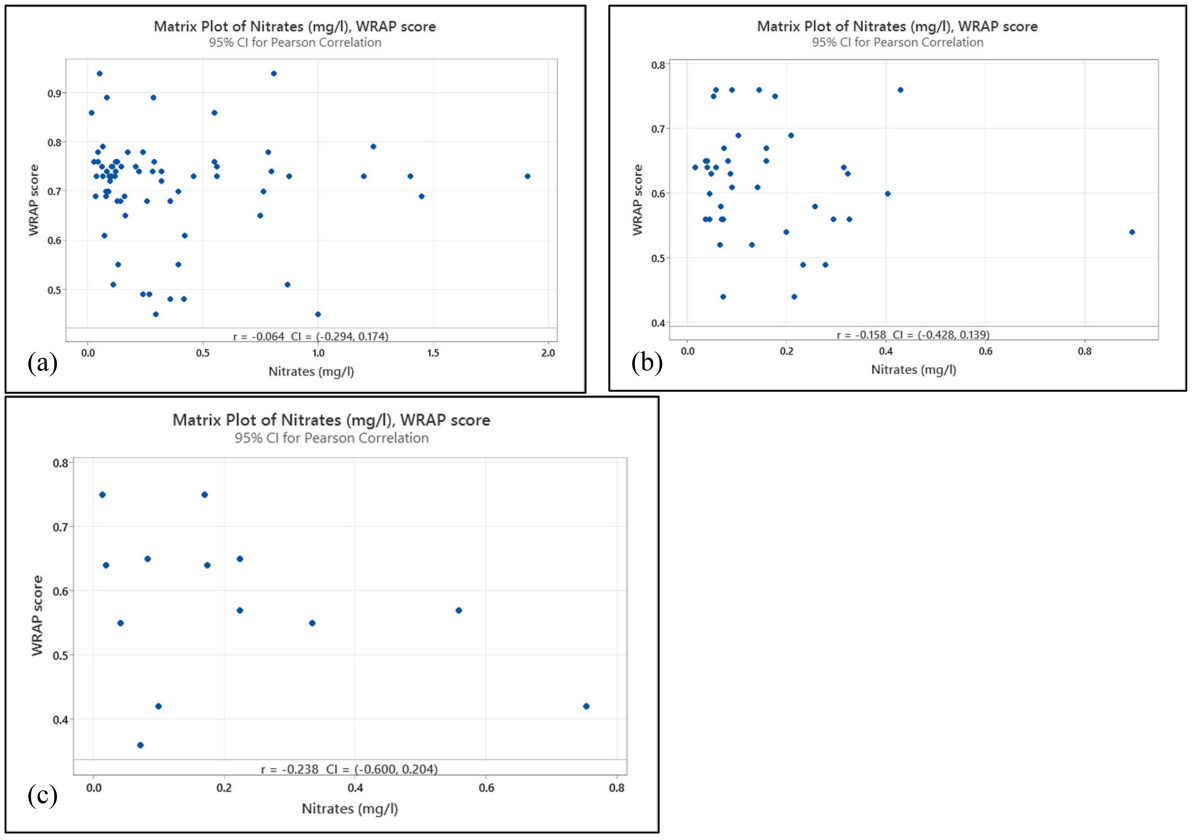


Figure 14. Pearson correlation plots between WRAP scores and nutrients (nitrates) in the water for Bigodi wetland system (a), Kahunge wetland system (b) and Businge wetland system (c).

conservation interventions, the apparent association between local conservation activities and the observed land-cover stability should be interpreted as an ecological association rather than proof of causation.

The potential importance of community-based management lies in the way it can connect conservation rules with local livelihoods. Conventional government-led protection may be constrained by limited enforcement capacity, insufficient field presence and weak integration of local resource users into decision-making. By contrast, community-based approaches may promote local ownership, incorporate local ecological knowledge, improve informal monitoring and make conservation benefits more visible to resource users. These mechanisms can increase the social legitimacy of conservation rules and potentially improve compliance. However, community-based management is not automatically effective; its success

depends on institutional legitimacy, equitable distribution of benefits, effective local leadership and continued support from government and conservation agencies. Thus, the relatively stable land-cover trajectory observed in Bigodi provides a strong basis for further investigation of community-based conservation, but does not by itself demonstrate that community management is universally more effective than government control (Kangalawe and Liwenga 2005; Douglas and Victor 2022).

The sanctuary also experienced an increase in woodlot cover (+3.17%), reflecting tree planting initiative by KAFRED, a common wetland restoration technique in participatory conservation programs (Finlayson et al. 2005) However, the ecological consequences of replacing natural wetland vegetation with woodlots depend on the species planted, their water requirements and their spatial position within the wetland (Finlayson and Woodroffe 1996). Consequently, increases in tree cover should not automatically be interpreted as equivalent to restoration of natural wetland functioning. Future assessments should distinguish between restoration of indigenous wetland vegetation and conversion to planted woody vegetation.

Kahunge exhibited an intermediate trajectory, characterized by a decline in wetland cover and an increase in grassland. This pattern may represent a transitional stage in wetland degradation. Repeated cultivation, grazing, vegetation clearance and alteration of surface drainage can reduce soil saturation and favor the expansion of grasses into areas that previously supported wetland vegetation. Such changes can progressively simplify habitat structure and reduce the capacity of the wetland to retain water and regulate nutrient and sediment movement (Acreman and Holden 2013; Douglas and Victor, 2022). The observed decline in small-scale agriculture alongside increasing grassland cover may reflect land-use transitions rather than ecological recovery. For example, previously cultivated areas may have been abandoned, become less productive, or shifted towards grazing or unmanaged grass-dominated vegetation. Therefore, a decline in agricultural cover should not automatically be interpreted as evidence of improved ecological condition.

In contrast, Businge wetland displayed severe degradation, with a reduction in wetland cover and substantial expansion of grasses and built-up areas. The doubling of built-up areas, though still low, points to unregulated settlement, possibly due to population pressure and weak enforcement of wetland protection laws. This is characteristic of peri-urban wetlands globally, where encroachment outpaces institutional responses (Gibbs 2016). Regionally in sub-Saharan Africa, high population growth and dependence on wetlands for subsistence farming have accelerated wetland losses. The pronounced loss in Businge likely reflects both direct anthropogenic conversion for seasonal grazing and agriculture, and indirect drivers such as drainage for cultivation, which alter hydrological regimes, encouraging terrestrial vegetation colonization. The continued dominance of small-scale agriculture further reflects the community's dependency on wetlands for subsistence. The shift toward grassland in KaWS and BuWS could also reflect climatic influences such as prolonged dry spells reducing inundation frequency, promoting colonization by grasses adapted to drier conditions. Without intervention, such transitions risk functional degradation-reduced water storage, filtration and biodiversity support.

The contrasting trajectories of the three wetland systems therefore suggest that wetland-use and wetland-cover change is influenced not only by the amount of human activity but also by the type, intensity and spatial distribution of that activity. Businge's stronger conversion may reflect cumulative effects of multiple pressures, whereas Bigodi's relative stability may reflect stronger conservation institutions and greater local incentives for maintaining wetland resources. This interpretation is consistent with studies showing that land-use change around protected areas is often driven by interacting demographic, economic and governance factors rather than by a single disturbance source (Southworth et al. 2010). The results therefore demonstrate that wetland conservation around Kibale National Park requires management approaches tailored to the specific pressures and governance conditions of individual wetland systems.

Spatial-temporal distribution of macrozoobenthos as indicators of habitat quality

Macrozoobenthos serve as vital biological indicators in wetland monitoring due to their long life cycles, relative immobility and differential sensitivity to pollutants and habitat changes (Rosenberg and Resh 2013). The diversity and distribution of macroinvertebrates reflect the cumulative impacts of physico-

chemical alterations, land-use changes and seasonal hydrological variability. In this study, the three wetland systems under study, Bigodi, Kahunge and Businge revealed important ecological gradients linked to their disturbance gradients. The differences in macrozoobenthic abundance and community composition among the three wetland systems suggest that biological responses were influenced by differences in habitat condition, environmental heterogeneity and disturbance history.

The dominant presence of Planorbidae and Ampullaridae, moderately tolerant mollusks, suggests a habitat with moderate organic enrichment and stable water chemistry and reflect the availability of vegetated and relatively stable aquatic habitats that provide food, attachment surfaces and refuge. In addition, the presence of rare and sensitive taxa like Baetidae, Elmidae, philopotamidae and Atyidae, though somewhat in low numbers, indicates the existence of microhabitats with high water quality and structural complexity, particularly in the sanctuary. Such ecological heterogeneity is often linked to intact riparian zones and vegetative buffers that moderate hydrology and nutrient loading (Dallas and Dallas 2016). In contrast, the dominance of Coenagrionidae in Businge, together with the lower overall abundance and greater contribution of relatively uncommon taxa, may indicate a more simplified or heterogeneous disturbance environment. However, family-level identification limits the extent to which the ecological tolerances of individual species can be inferred (Bailey et al. 2001). The results should therefore be interpreted as evidence of differences in community structure rather than definitive evidence that particular families are responsible for specific ecological processes.

The occurrence of pollution-tolerant taxa in disturbed locations is consistent with the general use of benthic macroinvertebrates as indicators of environmental condition (Bendary et al. 2023). Agricultural runoff, sedimentation, nutrient enrichment and hydrological alteration can modify benthic habitats and favor taxa capable of tolerating fluctuating environmental conditions (Masese et al. 2015). The relatively low diversity and strong dominance by Coenagrionidae in Businge may therefore reflect environmental filtering associated with habitat simplification and disturbance. Nevertheless, the observed community pattern may also be influenced by differences in substrate, aquatic vegetation and hydrological connectivity, which were not independently quantified in this study.

The higher diversity and ASPT values generally associated with better-quality sites support the usefulness of biological indicators for assessing ecological condition. However, diversity and ASPT measure partly different ecological properties. Diversity reflects the number and relative distribution of taxa, whereas ASPT is more strongly influenced by the environmental sensitivity of the taxa present. A disturbed site may therefore retain relatively high family richness while losing sensitive taxa and becoming dominated by tolerant groups. This explains why ASPT may provide a stronger signal of environmental stress than diversity alone in some wetland systems (Karr 1991; Hellawell, 2012). The particularly strong ASPT gradient observed in Businge is consistent with the greater landscape transformation and the more pronounced environmental stress indicated by the water-quality results.

Seasonal differences further demonstrate that macrozoobenthic communities respond not only to spatial disturbance but also to temporal changes in hydrology. Higher diversity during the dry season in Kahunge and Businge may be associated with greater habitat stability and reduced physical disturbance compared with periods of intense rainfall and runoff. Conversely, higher ASPT values during the wet season in some locations may reflect improved oxygenation, dilution of some pollutants, increased hydrological connectivity or the movement of organisms between connected habitats. These contrasting responses demonstrate that seasonal effects can influence different biological metrics in different ways. Consequently, a single-season assessment may provide an incomplete representation of wetland biological condition, particularly in systems with strong seasonal hydrological variation (Dudgeon 2000; Datry 2012).

Physico-chemical parameters of wetland waters along the disturbance gradient

The physico-chemical results indicate that the three wetlands did not exhibit a simple and consistent gradient in which all water-quality parameters deteriorated progressively from high- to low-quality sites. Instead, the results suggest that water chemistry was shaped by a combination of local human activities, hydrological connectivity, seasonality and wetland-specific processes. This finding is important because the absence of a consistent chemical gradient does not necessarily imply the absence of ecological disturbance.

Different pollutants may enter wetlands through different pathways, while hydrological mixing can distribute pollutants beyond their immediate sources (Chapman 1996; Namaalwa et al. 2020).

The particularly low dissolved oxygen concentrations observed at some low-quality sites in Businge are ecologically important because oxygen availability directly influences aerobic aquatic organisms and the biogeochemical processes occurring in wetland sediments (Michael et al. 2023). Low dissolved oxygen may result from the decomposition of organic matter, nutrient enrichment and high microbial oxygen demand. Such conditions can reduce the suitability of habitats for oxygen-sensitive organisms and may help explain the poorer biological condition observed in parts of Businge (Namaalwa et al. 2020).

The seasonal increases in nitrate and phosphate, particularly in Businge, are consistent with the possibility of nutrient inputs from agricultural activities, livestock and other human activities within or adjacent to the wetland. In agricultural landscapes, nutrients may enter wetlands through surface runoff, soil erosion and waste deposition (Skagen et al. 2008). However, because this study did not directly trace nutrient sources, these processes should be regarded as plausible explanations rather than demonstrated source pathways. Similar relationships between agricultural land use, nutrient enrichment and wetland water-quality changes have been documented in Ugandan wetland systems (Mucunguzi 2007; Twesigye et al. 2011; Namaalwa et al. 2020).

The PCA further demonstrated that the wetlands had distinct chemical signatures that varied between seasons. The separation of Businge during the dry season in association with relatively higher nitrate, hardness and pH suggests that its water chemistry was influenced by a different combination of environmental processes from those operating in Bigodi and Kahunge. During the wet season, the greater separation of Bigodi and the association of some parameters with the principal components indicate that rainfall and hydrological inputs altered the chemical characteristics of the wetlands. These results suggest that seasonality may be at least as important as the broad WRAP disturbance category in shaping water chemistry during the sampling period.

Relationship between WRAP scores and biological indicators

The weak positive relationships between WRAP scores and macrozoobenthic diversity should not be interpreted simply as evidence that qualitative assessment is inadequate. Several ecological and methodological factors may explain the weak associations. First, WRAP provides a rapid, composite assessment of visible site characteristics, whereas macrozoobenthic communities respond to a broader range of environmental conditions, including substrate composition, dissolved oxygen, nutrient concentrations, hydrological connectivity, flow variability, aquatic vegetation and microhabitat availability (Schwingshackl et al. 2026). These variables may vary substantially within a site and may not be fully captured by a rapid visual assessment. Second, sites with similar WRAP scores may experience different types of disturbance. For example, agricultural cultivation, livestock access, vegetation clearance, settlement, waste disposal and channel modification may produce similar visual evidence of degradation but may affect macrozoobenthos through different ecological pathways. A score that combines several disturbance attributes into a single value may therefore obscure important differences in the type and intensity of stress experienced by aquatic communities. Third, the temporal scale of the two assessments may not be identical. WRAP scores represent the condition observed during the field assessment (Miller 1999), whereas macrozoobenthic communities integrate environmental conditions over a longer period through their life cycles, recruitment and re-colonization processes. A short-term improvement or deterioration in local conditions may therefore not immediately produce a corresponding biological response. Seasonal changes in water flow and habitat availability may further weaken the relationship between a single rapid assessment score and biological indicators (Dudgeon 2000; Datry 2012). Fourth, spatial heterogeneity within the wetland systems may have reduced the strength of the correlations. Macrozoobenthos were sampled from particular habitats and sampling points, whereas WRAP scores represented broader site characteristics. Consequently, a mismatch between the spatial scale of the habitat assessment and the biological sampling could weaken the observed relationship. Finally, the statistical power of the correlations was constrained by the number and distribution of sampling sites across the quality categories. The unequal representation of high-, medium- and low-quality sites, particularly where some categories contained relatively few sampling locations, limits the strength with which relationships can be detected. This is especially important when

interpreting weak correlations, because a weak coefficient may reflect either a genuinely weak ecological relationship or limited statistical power. The results therefore support the use of WRAP as a rapid screening and site-classification tool, but not as a stand-alone measure of ecological integrity.

The weak WRAP-biological relationships consequently reinforce the importance of integrating multiple indicators rather than relying on a single assessment approach. A more comprehensive assessment of wetland ecological condition should combine landscape change, habitat structure, biological communities and water chemistry. Such an approach is consistent with the concept of biological integrity, which emphasizes the need to evaluate ecological condition through multiple dimensions of ecosystem structure and function rather than through a single environmental variable (Karr 1991; Kansiime et al. 2007; Kentula and Paulsen 2019). In the present study, the combined interpretation of wetland-cover change, macrozoobenthic indicators and physico-chemical conditions provided a more informative assessment of ecological condition than any one of these indicators could provide independently.

Overall, the results suggest that wetland ecological integrity around Kibale National Park is shaped by interacting pressures rather than by a single disturbance gradient. Effective conservation should therefore combine protection of remaining wetland habitat, control of agricultural and settlement encroachment, maintenance of hydrological connectivity, community participation and repeated biological and water-quality monitoring. Importantly, the present study provides evidence of associations among wetland-cover change, biological condition and water chemistry, but longitudinal and experimental studies will be required to establish causal relationships between specific management interventions and ecological recovery.

Conclusions

The findings of this study underscore the intricate and dynamic interplay between anthropogenic/disturbance pressures and wetland ecological integrity in the three wetland systems around KNP and within Kamwenge district: Bigodi, Kahunge and Businge wetlands. Over the 9-year period from 2016 to 2025, Wetland-use and wetland-cover changes demonstrated distinct trajectories, reflecting differential levels of protection, community involvement and wetland-use pressures. Bigodi wetland, particularly the sanctuary managed under KAFRED, maintained ecological stability and high wetland cover, illustrating the effectiveness of community-led conservation. In contrast, Kahunge and Businge wetlands experienced pronounced degradation, particularly in the form of wetland loss, very high levels of small-scale agriculture and expansion of grasses and built-up areas.

The sensitivity of macrozoobenthos to disturbance revealed spatial-temporal variations in macrozoobenthos distribution across the disturbance gradients and seasons. ASPT and family diversity, particularly in KaWS and BuWS showed seasonally significant differences in macrozoobenthos spatial-temporal distribution. Bigodi wetland system again emerged as the most biologically diverse, while Businge wetland system exhibited signs of ecological stress. The sensitivity of macrozoobenthos to disturbance was confirmed through their distribution patterns across the disturbance gradients and seasons. Furthermore, ASPT was a more robust indicator of environmental stress than diversity alone, particularly in Businge wetland system where significant differences were observed.

Physico-chemical water quality assessments revealed that most parameters generally fell within NEMA standards, with few exceptions such as low DO and seasonal nitrate, nitrite and phosphate spikes. These deviations, particularly in Businge, were indicative of anthropogenic stress. The correlation between WRAP scores and biotic indicators was weak, highlighting the necessity of an integrative approach for accurate wetland assessment.

Overall, this study affirms the need for multi-indicator monitoring frameworks that consider land-use change, biodiversity and water chemistry to effectively assess and manage wetland ecological integrity.

Recommendations

i. Enhance Community-Based Wetland Management; the positive ecological outcomes observed in Bigodi Wetland system and sanctuary in particular affirm the value of community stewardship. This model should be replicated in Kahunge and Businge and other wetland ecosystems in Uganda, with

targeted training, capacity building and incentive-based conservation schemes to engage local stakeholders.

ii. Strengthen wetland governance and policy enforcement, national and district authorities must enforce wetland protection regulations under Uganda's Wetlands Policy and the Environment Act. Illegal encroachment and degradation, especially in Businge wetland system, require decisive legal and administrative action.

iii. Establish Integrated Wetland Monitoring Programs; Monitoring should combine rapid wetland assessments (WRAP), biological indices (ASPT and diversity) and chemical analyzes to provide a holistic view of ecological status. Longitudinal monitoring, particularly across seasons, is critical for capturing wetland dynamics.

iv. Invest in wetland restoration programs; priority should be given to rehabilitating degraded wetland systems such as Businge. Restoration interventions may include re-vegetation with indigenous wetland species.

Areas for further research

- Long-term monitoring of wetland ecosystem trajectories under climate variability and land-use change, particularly focusing on hydrological alterations and invasive species in wetlands around protected areas and others.
- Experimental studies on the resilience and recovery of macrozoobenthos communities following wetland restoration in wetland systems around protected areas and others.
- Socio-economic valuation of wetland ecosystem services comparing protected vs unprotected wetlands to inform cost-benefit analysis for policy design.

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