Journal of Food Health and Bioenvironmental Science (January - April 2024), 17(1): 11-22

Journal of Food Health and Bioenvironmental Science

Journal homepage : http://jfhb.dusit.ac.th/

Diversity and Structure of Aquatic Insect Communities in Relation to Water Quality Parameters in The Kasetsart University Drainage Ditches, Central Thailand

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Article info

Article history:
Received : 2 March 2024
Revised : 4 April 2024
Accepted : 17 April 2024

Keywords:
Benthic macroinvertebrate, Biodiversity, Drainage ditches, Water quality

Abstract

This study evaluates the impact of anthropogenic influence on the drainage ditches of Kasetsart University's Kamphaeng Saen Campus in Nakhon Pathom Province, using water quality measurements and aquatic insect data collected during a six-month period from October 2022 to March 2023. Six sampling ditches with various human activities were chosen. Organic wastes from agriculture, residences and buildings were the primary causes of pollution. A total of 6,232 aquatic insects from six orders—Hemiptera (28.32%), Coleoptera (7.76%), Odonata (12.52%), Diptera (36.33%), Ephemeroptera (2.47%) and Trichoptera (12.6%)—were collected. In the current study, the order Diptera was the most diversified and relatively abundant in all sampling sites. However, the order Trichoptera was most common in KUKPS6, a running water ditch. The orders Hemiptera and Coleoptera had the highest abundance in the KUKPS1, while Odonata and Ephemeroptera had the highest abundance in KUKPS2. The Shannon-Weiner diversity index for aquatic insects was between 1.527 and 2.959, indicating slightly good water quality. According to the Canonical Correspondence Analysis (CCA), dissolved oxygen, orthophosphate, nitrate-nitrogen, turbidity and pH were the most influential factors. Based on the data on physicochemical water quality and entomofauna composition, the results concluded that Kasetsart University's drainage ditches were slightly polluted.

Introduction

Water contamination is one of the most important environmental issues confronting Thailand today. The majority of sewage enters waterways without being properly treated, reducing the quality of Thailand's freshwater supply. Aquatic assessments have traditionally focused on physical and chemical variables; however, the recent inclusion of biota in monitoring programs has proven to be a more responsive and sensitive measure of environmental conditions (Bonada et al., 2006). In the past and until now, physicochemical measurements have

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typically been used to evaluate the water quality in freshwater ecosystems. However, these measurements do not yield ecological information on their own because it may be difficult to fully assess the synergistic effects of pollution on aquatic biotic communities (Arimoro et al., 2015). Macroinvertebrates are an essential part of ecosystems among the aquatic biota because they provide a vital intermediate route for moving and using matter and energy. Furthermore, individual taxa respond differently to various pollutants and can provide information about water quality over time (Bonada et al., 2006; Odume et al., 2012).

Aquatic insects are diverse and can be found in a range of freshwater environments, including lotic (running water) and lentic (standing water) (Williams & Williams, 2017). Aquatic insects are arthropods that live or spend most of their lives in water (Arimoro & Ikomi, 2008; Williams & Williams, 2017). They are of great significance to the aquatic habitat where they are found; therefore, their existence in water bodies serves various of purposes. While some serve as food for fish and other invertebrates, others are vectors of disease pathogens to humans and animals (Iwamoto et al., 2022). Most significantly, aquatic insects are excellent indicators of water quality because they tolerate a wide range of environmental disturbances (Arimoro & Ikomi, 2008). Some are extremely vulnerable and sensitive to pollution, while others can survive and thrive in highly contaminated waters (Hepp et al., 2013). Anthropogenic activities included mining, runoff from agricultural fields, releases from home sewage and laundry into waterways. As a result, most bodies of water have become increasingly polluted, significantly affecting their quality and health. This alters the physicochemical qualities of water, including temperature, dissolved oxygen, alkalinity, phosphates, nitrates and metal concentrations. Variations in these water qualities have a significant impact on the distribution patterns of aquatic insects in the water, since some are extremely sensitive to pollution while others are moderately tolerant or completely tolerant of pollution and environmental disturbances (Hepp et al., 2013).

Many approaches have been developed to examine water quality issues. Physical parameters (such as stream bank erosion, turbidity, sedimentation, siltation, flow patterns, water temperature, riparian cover and debris) and chemical parameters (such as dissolved oxygen, biochemical oxygen demand, pH, alkalinity, hardness, nutrients, metals, and organic compounds) have been used to assess water quality (Aweng et al., 2011; Zarei & Bilondi, 2013). Unfortunately, because sampling and analysis are expensive and pollutant concentrations fluctuate widely with location and time, physical and chemical monitoring alone frequently fail to detect non-point-source pollution concerns. In contrast, biological monitoring provides information about both historical and present circumstances (Suhaila et al., 2014). The additional integration of biological parameters into physicochemical assessments has proven to be a more complete method to fully assess pollutant effects in aquatic ecosystems, particularly in lotic systems (Prommi & Payakka, 2015). Bioassessment provides more reliability in evaluating the presence and impact of pollutants because lotic systems are subjected to flushing during storm events and contaminants may be swept away without any apparent effect (Borisko et al., 2007; Buss & Vitorino, 2010). Benthic macroinvertebrates are the most significant biota utilized in bioassessment investigations. Benthic macroinvertebrates are commonly used as biological indicators in river quality assessments because they are influenced by changes in river natural variables such as width, depth, substratum type, water velocity and physicochemical variables caused by both natural and human activities (Prommi & Payakka, 2015; Xu et al., 2014).

Aquatic insects are one of the most commonly used benthic macroinvertebrate groups for assessing the health of aquatic ecosystems (Prommi & Payakka, 2015; Xu et al., 2014). They are excellent indicators because they represent a diverse group of long-lived sedentary species that respond strongly and reliably to human influences on aquatic systems (Williams & Williams, 2017). Some aquatic insects respond to specific changes in water conditions and aquatic ecologists use them as markers of river health. The presence or absence of a specific aquatic insect indicates the level of pollution, though the exact causative physicochemical pollutant can be detected using physicochemical methods (Prommi & Payakka, 2015). For various reasons, aquatic insects are critical to the health of rivers and streams and can thus be considered surrogates for river and stream wealth. First, logistically, they make suitable study specimens since they are abundant, easily surveyed and taxonomically diverse. The diversity and quantity of aquatic insects indicate the general health of a body of water. Identifying the variety and community composition of a sample of aquatic insects in specific water bodies can help establish the richness and abundance of the aquatic insect fauna in that stream. Usually, research and management of lentic aquatic ecosystems in lowland settings has been concentrated on lakes or ponds, ignoring minor aquatic habitats created by humans, such as drainage ditches. In lowland agricultural areas, drainage ditches are typical building features. These man-made linear water bodies are linked together to build networks that can provide valuable habitats for many water-associated organisms. The objectives of this study were to inventory aquatic insect diversity in drainage ditches as well as examine the water quality status of Kasetsart University's drainage ditches and how this affects aquatic insects' diversity and richness.

Materials and methods

1. Study area and sampling stations

This study was conducted at Kasetsart University's drainage ditches. Six drainage ditches (KUKPS1, KUKPS2, KUKPS3, KUKPS4, KUKPS5 and KUKPS6) (Fig. 1) were selected for the study based on the material used to construct the ditch. The sampling ditches KUKPS2, KUKPS4 and KUKPS6 are concrete irrigation canals that drain agricultural water supplies. There is no marginal vegetation. The substratum is composed of mud, detritus and some gravel. The sampling ditches KUKPS1, KUKPS3 and KUKPS5 are drainage ditches that gather wastewater from agricultural and residential areas. There is marginal vegetation and water-floating plants. The substratum is predominantly loamy and clay-based.

2. Water sampling for physical and chemical variables

Water samples were collected monthly at each station for six months, from October 2022 to March 2023. Three replicates of selected physicochemical water quality parameters were recorded directly at the sampling sites: pH using portable pH, water temperature (WT, °C), dissolved oxygen (DO, mg/L) using EcoScan DO 110, total dissolved solids (TDS, mg/L) and electrical conductivity (EC, µS/cm) using CyberScan CON 11. Three replicate water samples from each site and date were collected in 500-mL polyethylene bottles and brought to the laboratory in an ice-filled cooler box.

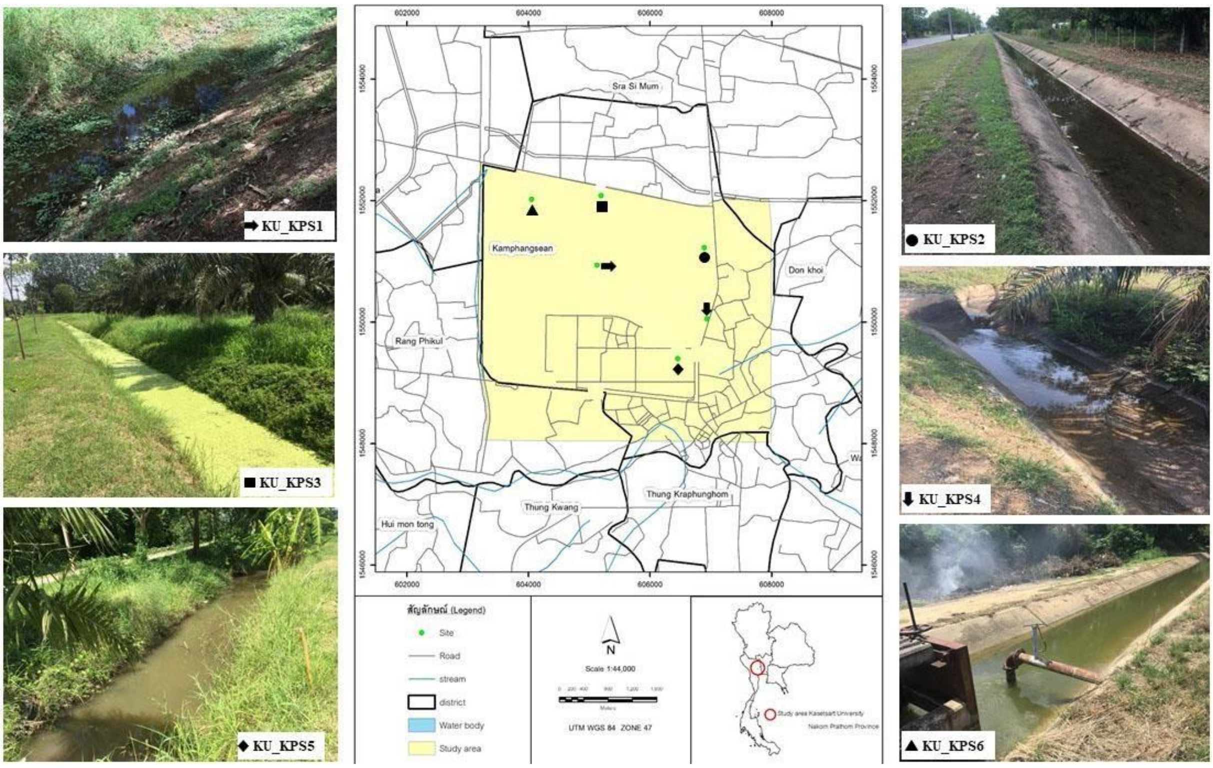

Fig. 1 Map of Kasetsart University, Kamphaeng Saen Campus, Nakhon Pathom Province, Thailand, showing the location of the drainage ditches, KUKPS1–KUKPS6

In the laboratory, ammonia-nitrogen (NH_3-N, mg/L) was measured by the Nessler method, using a spectrophotometer (DR/2010 model 49300-00; Hach, USA); nitrate-nitrogen (NO_3-N, mg/L) by the cadmium reduction method, using a spectrophotometer (DR/2010 model 49300-00; Hach, USA); orthophosphate (PO_4^{3-}, mg/L) by the ascorbic acid method, using a spectrophotometer (DR/2010 model 49300-00; Hach, USA); and turbidity (TUB, NTU) using a spectrophotometer (DR/2010 model 49300-00; Hach, USA) (APHA, 1992).

3. Aquatic insect sampling and processing

Monthly sampling was carried out from October 2022 to March 2023. Aquatic insects were sampled using an aquatic D-hand net with a 30 x 30 cm frame, 250 μm mesh and 50 cm length throughout the sampling. At each sampling location, samples were collected from the ditches' margin and bottom across a 3-meter stretch. The sample time at each habitat was three minutes. In each sampling interval, three replicate samples were taken at each location. The samples were placed on white trays for sorting and screening for aquatic insects. The aquatic insects were hand-picked from the tray. Any non-aquatic insects captured were promptly released into the ditches. The samples were moved to properly labeled plastic containers, stored in 80% ethanol and returned to the laboratory for examination. In the laboratory, aquatic insects were sorted, counted and identified using a stereoscopic microscope. Aquatic insect species were identified to the lowest taxonomic level using available keys (Dudgeon, 1999; Yule & Yong, 2004) and assistance from aquatic insect taxonomists and experts. All sorted samples were stored in appropriately labeled vials containing 70% ethanol. The voucher material was deposited at the Faculty of Liberal Arts and Science, Kasetsart University, Kamphaeng Saen Campus, Thailand.

4. Data analyses

The mean and standard deviation of each physicochemical variable were computed for each station. Using IBM SPSS Version 20.0, a one-way ANOVA with Tukey's (HSD) post hoc test was used to compare physicochemical characteristics between sampling dates and sampling sites. The four community indices (richness, evenness, Shannon diversity and Simpson diversity) were calculated. Species richness (S) was the number of species in each sample, with any doubtful identifications (early instar, damaged) that could not be verified as new taxa excluded from the total. Evenness, Shannon diversity and Simpson diversity were calculated with PC-ORD 5.1 (McCune & Mefford, 2006). Using PC-ORD version 5.1, canonical correspondence analysis (CCA) was utilized to evaluate the relationships between aquatic insects and environmental variables. Cluster analysis and non-metric multidimensional scaling (NMDS) were used to identify sampling sites based on aquatic insects using Ward's linkage approach with Euclidean distance measures in PC-ORD software.

Results and discussion

1. Environmental variables in drainage ditches

Table 1 summarizes the mean and standard deviation of the environmental variables at each sampling location. The ammonia-nitrogen and nitrate-nitrogen did not vary significantly between the sampling sites ($p > 0.05$) and the dissolved oxygen, pH, total dissolved solids, electrical conductivity and nitrate-nitrogen did not vary significantly during the sampling periods ($p > 0.05$), using ANOVA (Table 1).

Water temperatures in all sampling sites ranged from 27.06±0.39°C to 30.97±0.34°C. KUKPS4 had the highest temperature, whereas KUKPS1 had the lowest temperature. Many factors influence water temperature fluctuations, including sample timing and habitat conditions. Temperatures were lower during the cold-dry season (December to January) compared to the hot-dry season (March). Tropical waters require normal minimum and maximum temperatures (27.06±0.39°C and 30.97±0.34ºC, respectively) for aquatic organisms to grow.

The dissolved oxygen levels differed significantly among the drainage ditch locations where the aquatic insects were gathered. The greatest DO was reported in KUPKS6 (8.04±0.32 mg/L), followed by KUKPS4 (7.31±0.19 mg/L) and KUKPS2 (6.94±0.42 mg/L), which were the sites with running water. The lowest (1.89±0.29, 1.89±0.28 and 2.92±0.44 mg/L) were found in KUKPS1, KUKPS3 and KUKPS5, where lentic water was present. Aquatic insects were found in drainage ditch samples, with DO levels ranging from 1.89±0.29 mg/L to 8.04±0.32 mg/L. The atmosphere and photosynthesis are sources of dissolved oxygen in aquatic environments and its concentration is determined by its solubility (which decreases with increasing water temperature), whereas respiration of submerged plants and animals and aerobic bacteria, including their metabolic activity in decomposing dead organic matter, reduces dissolved oxygen (Gupta & Gupta, 2006).

Table 1 Physicochemical parameters at sampling sites (KUKPS1–KUKPS6)

Factor/Site	**KUKPS1**	**KUKPS2**	**KUKPS3**	**KUKPS4**	**KUKPS5**	**KUKPS6**
WT (ºC)	27.28 ± 0.44^{a}	31.10 ± 0.51^{c}	28.89 ± 0.51^{ab}	31.11 ± 0.39^{c}	28.58 ± 0.57^{ab}	30.56 ± 0.43^{bc}
DO (mg/L)	1.89 ± 0.29^{a}	6.94 ± 0.42^{b}	1.89 ± 0.28^{a}	7.31 ± 0.19^{b}	2.92 ± 0.44^{a}	8.04 ± 0.32^{b}
pH	7.34 ± 0.10^{a}	8.68 ± 0.09^{b}	7.58 ± 0.01^{a}	8.62 ± 0.05^{b}	7.64 ± 0.05^{a}	8.85 ± 0.11^{b}
TDS (mg/L)	137.55 ± 8.52^{ab}	140.67 ± 9.16^{ab}	654.00 ± 18.99^{c}	96.70 ± 2.01^{a}	209.12 ± 13.78^{b}	95.46 ± 2.50^{a}
EC (µS/cm)	272.28 ± 17.20^{a}	199.05 ± 7.22^{a}	892.01 ± 73.11^{b}	192.42 ± 3.85^{a}	417.37 ± 27.22^{a}	190.33 ± 5.06^{a}
TUR (NTU)	37.50 ± 3.41^{c}	39.00 ± 6.34^{c}	22.78 ± 2.16^{ab}	19.83 ± 1.93^{ab}	29.16 ± 1.89^{bc}	12.83 ± 0.91^{a}
NH_3-N (mg/L)	0.87 ± 0.22^{a}	1.12 ± 0.45^{a}	0.63 ± 0.09^{a}	0.81 ± 0.13^{a}	0.54 ± 0.09^{a}	0.84 ± 0.56^{a}
NO_3-N (mg/L)	1.89 ± 0.20^{a}	1.98 ± 0.48^{a}	1.16 ± 0.09^{a}	1.25 ± 0.08^{a}	1.37 ± 0.13^{a}	1.4 ± 0.15^{a}
PO_4^{3-} (mg/L)	0.52 ± 0.05^{ab}	0.69 ± 0.19^{ab}	0.93 ± 0.16^{b}	0.29 ± 0.03^{a}	0.51 ± 0.06^{ab}	0.3 ± 0.07^{a}
Factor/Month	**October**	**November**	**December**	**January**	**February**	**March**
WT (ºC)	30.08 ± 0.45^{b}	30.76 ± 0.55^{b}	27.06 ± 0.39^{a}	27.87 ± 0.60^{a}	30.79 ± 0.36^{b}	30.97 ± 0.34^{b}
DO (mg/L)	6.43 ± 0.38^{a}	4.95 ± 0.82^{a}	5.12 ± 0.85^{a}	4.63 ± 0.80^{a}	3.74 ± 0.51^{a}	4.13 ± 0.65^{a}
pH	7.93 ± 0.14^{a}	8.26 ± 0.18^{a}	8.30 ± 0.17^{a}	8.23 ± 0.16^{a}	8.11 ± 0.14^{a}	7.90 ± 0.17^{a}
TDS (mg/L)	216.82 ± 51.72^{a}	234.22 ± 58.48^{a}	200.02 ± 41.68^{a}	224.11 ± 39.17^{a}	209.34 ± 53.57^{a}	248.99 ± 63.51^{a}
EC (µS/cm)	433.17 ± 82.35^{a}	469.58 ± 87.10^{a}	400.88 ± 83.61^{a}	448.22 ± 78.33^{a}	206.47 ± 13.28^{a}	205.15 ± 20.68^{a}
TUR (NTU)	17.38 ± 2.68^{a}	23.44 ± 2.28^{a}	27.72 ± 2.19^{ab}	28.77 ± 2.07^{ab}	21.83 ± 2.27^{a}	41.94 ± 6.93^{b}
NH_3-N (mg/L)	0.47 ± 0.04^{a}	0.51 ± 0.06^{a}	0.51 ± 0.04^{a}	0.49 ± 0.04^{a}	0.64 ± 0.15^{a}	2.19 ± 0.38^{b}
NO_3-N (mg/L)	1.81 ± 0.20^{a}	1.16 ± 0.15^{a}	1.27 ± 0.11^{a}	1.27 ± 0.09^{a}	1.63 ± 0.10^{a}	2.07 ± 0.47^{a}
PO_4^{3-} (mg/L)	0.58 ± 0.07^{ab}	0.51 ± 0.07^{ab}	0.29 ± 0.04^{a}	0.36 ± 0.04^{a}	0.64 ± 0.14^{ab}	0.87 ± 0.22^{b}

Remark: The mean and standard deviation for each site (above) and the mean and standard deviation for each sampling date measured from October 2022 to March 2023 (below). WT = water temperature (°C), DO = dissolved oxygen (mg/L), EC = electrical conductivity (µS/cm) and TDS = total dissolved solids (mg/L); turbidity is measured in NTU, SO_4^{2-} and NO_3-N and PO_4^{3-} and NH_3-N are measured in mg/L. Values followed by the same letter are not significantly different ($p>0.05$).

Water pH varied significantly among habitats, ranging from 7.34 to 8.85. The drainage ditch KUKPS6 had the highest mean value (8.85±0.11) and KUKPS1 had the lowest (7.34±0.10). Natural shallow-water environments contain high pH values, which have a direct impact on photosynthesis that cannot be explained by a lack of inorganic carbon (Middelboe & Hansen, 2007). The pH decreased throughout the rainy season. This could be due to the increased organic waste passed into the waterway by surface runoff during the wet season, which tends to diminish dissolved oxygen through organic decomposition, hence lowering pH.

Electrical conductivity values varied significantly between sites, with a mean of 190.33±5.06 µS/cm for the KUKPS6 and 892.01±73.11 µS/cm for the KUKPS3. The KUKPS5 and KUKPS1 sites had relatively high mean electrical conductivity levels of 417.37±27.22 µS/cm and 272.28±17.20 µS/cm, respectively. Turbidity, total dissolved solids and electrical conductivity were found to be elevated in KUKPS1, KUKPS3 and KUKPS5, most likely due to dissolved particle accumulation (Dida et al., 2015). This study found that electrical conductivity decreased during the dry season compared to the wet season. Low precipitation, higher air temperatures, increased evapotranspiration rates and total ionic concentration and saltwater intrusions from subsurface sources could all cause electrical conductivity to rise. It could potentially result from rapid decomposition and mineralization by bacteria and nutrient regeneration from bottom sediments (Herburt et al., 2015). Electrical conductivity and total dissolved solids may also be increased due to water contamination caused by agricultural activities and industrial effluent, which decreases water quality (Rey-Romero et al., 2022). The greater turbidity appeared during the rainy season, which could have been caused by significant rainfall. The increase in suspended solids obstructed light, increasing turbidity. Turbidity has several negative effects on freshwater, including decreased light penetration, which reduces primary and secondary production; increased adsorption of nutrient molecules to suspended materials, making nutrients unavailable for plankton production; decreased oxygen concentration; and clogged filter-feeding apparatus and digestive organs of planktonic organisms, which may affect larval production (Gupta & Gupta, 2006).

The concentrations of dissolved nutrients (NH_3-N, NO_3-N and PO_4^{3-}) ranged from 0.54±0.09 mg/L (KUKPS5) to 1.12±0.45 mg/L (KUKPS2), 1.16±0.09 mg/L (KUKPS3) to 1.98±0.48 mg/L (KUKPS2) and 0.29±0.03 mg/L (KUKPS4) to 0.93 ± 0.16 mg/L (KUKPS3). The increased phosphate concentration in drainage ditch habitats could be attributed to land use management practices. Other major sources of phosphorus in freshwater are atmospheric precipitation, geochemical conditions and groundwater (Lawniczak et al., 2016). The majority of nitrogen in natural aerobic water occurs as nitrates in different proportions depending on the nature of the watershed, seasons, pollution level and plankton population (Hong et al., 2023).

2. Aquatic insect communities

This study collected a total of 6,232 aquatic insects from 26 families and 44 genera across 6 orders (Fig. 2, Table 2). Diptera (2,264 individuals, 36.21%), Hemiptera (1,785 individuals, 28.55%), Trichoptera (785 individuals, 12.56%), Odonata (780 individuals, 12.48%), Coleoptera (484 individuals, 7.74%) and Ephemeroptera (154 individuals, 246%) were the most common orders found in the university's six drainage ditches (Fig. 2, Table 2). Furthermore, Chironomus spp. (30.8%) was the most common taxon, followed by *Amphipsyche meridiana* (12.6%), *Cratilla* sp. (8.39%), *Diplonychus indicus* (5.47%) and *Diplonychus rusticus* (5.09%). However, the most prevalent taxa for each drainage were *Chironomus* spp., KU_PKS1 (20.09%), KUKPS2 (31.38%), KUKPS3 (21.85%), KUKPS4 (59.77%) and KUKPS5 (18.23%). Except for KUKPS6, the main taxon was *Amphipsyche meridiana* (38.99%) (Table 2).

Diptera was the dominant order with the most species, comprising five families and accounting for 36.33% of all aquatic insects collected. The family Chironomidae and the species *Chironomus* spp. had the highest abundances among all Diptera families and species. Chironomidae are the most successful aquatic insect taxa, inhabiting all freshwater bodies, including contaminated and eutrophic ones (Grzybkowska et al., 2020). One of the main reasons for the Chironomidae's abundance is their diverse feeding patterns and food preferences (Antczak-Orlewska et al., 2021). The larval abundance of Culicidae, *Aedes* sp., and *Culex* sp. was low at all sites due to the number of mosquito larvae and predators (Reunura & Prommi, 2020). According to Kweka et al. (2012), higher grass cover reduces sunlight penetration into the environment, affecting algal biomass photosynthetic efficiency and other aquatic forms, which serve as additional food sources for mosquito larvae. Other researchers have discovered that grass cover affects mosquito oviposition site selection, resulting in a direct effect on larvae abundance (Mala et al., 2011; Bashar, 2016).

Hemiptera found in this study had the second-highest abundance of aquatic insects. Ten families were registered in Hemiptera, accounting for 28.33% and being dominated by the Belostomatidae and Notonectidae families. Belostomatidae, *Diplonychus indicus* and *Diplonychus rusticus* had the highest Hemipteran families or species abundance. *Diplonychus* was the most common genus found in nearly all sampling ditches and during the sampling period, particularly in lentic habitats (KUKPS1, KUKPS3 and KUKPS5). Belostomatidae are generally top predators in situations without vertebrates due to their relatively large size (Ohba, 2019). It is commonly found in both temporary and permanent water basins (Reunura & Prommi, 2020). Notonectidae came next, with 312 individuals. Hemipterans are thought to be excellent predators of freshwater snails and mosquito larvae in aquatic environments (Dida et al., 2015). Notonectids are also known to be voracious feeders of mosquito larvae (Dida et al., 2015). Mahenge et al. (2023) concluded that notonectid predators have the capacity to impact mosquito populations through direct or indirect effects. Direct evidence of notonectid predation on mosquito larvae was eventually discovered, confirming their primary role in mosquito larval control.

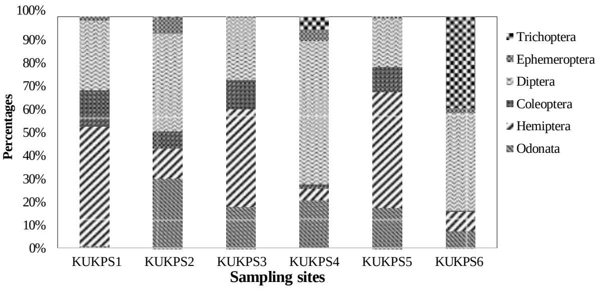

Fig. 2 Distribution and abundance of selected groups of aquatic insects in the sampling stations of Kasetsart University's drainage ditches during the study periods (October 2022 to March 2023)

Table 2 Diversity of aquatic insects in Kasetsart University's drainage ditches from October 2022 to March 2023 with Dominance Status of different species of aquatic insects in the University's drainage ditches according to Engelmann's scale (1978)

Order/Family	**Species**	**Abbr.**	**Sampling site**						**Total**	**%RA***	**Dominance Status**
			KUKPS1	**KUKPS2**	**KUKPS3**	**KUKPS4**	**KUKPS5**	**KUKPS6**			
Odonata											
Coenagrionidae	*Enallagma* sp.	Ena	5	65	11	13	25	31	150	2.41	recedent
	Telebasis sp.	Tel	3				52		55	0.88	subrecedent
Gomphidae	*Ictinogomphus* sp.	Ict			2	4	7	3	16	0.26	subrecedent
Libellulidae	*Hydrobasileus* sp.	Hyb		36					36	0.58	subrecedent
	Cratilla sp.	Cra	6	133	103	91	77	113	523	8.39	subdominant
Hemiptera											
Belostomatidae	*Diplonychus indicus*	Din	110	5	105		121		341	5.47	subdominant
	Diplonychus rusticus	Dru	84	3	133		96	1	317	5.09	subdominant
Gerridae	*Amemboa* sp.	Ame						5	5	0.08	subrecedent
	Limnogonus nitidus	Lni	21	73	4	27	1	50	176	2.82	recedent
	Limnogonus sp.	Lim	5	3					8	0.13	subrecedent
	Ventidius sp.	Ven				1		103	104	1.67	recedent
Helotrephidae	*Hydrotrephes yangae*	Hya	63	2	2		45	1	113	1.81	recedent
Hydrometridae	*Hydrometra annamana*	Han	2		3		1		6	0.1	subrecedent
	Hydrometra greeni	Hgr	57		2		7		66	1.06	subrecedent
Mesoveliidae	*Mesovelia horvathai*	Mho	58		4				62	0.99	subrecedent
	Mesovelia vittegera	Mvi	19		1		20		40	0.64	subrecedent
Micronectidae	*Micronecta quadristrigata*	Mqu	2	6	2		15		25	0.4	subrecedent
Nepidae	*Ranata* sp.	Ran	13		1		9	1	24	0.39	subrecedent
	Cercotmetus asiaticus	Cas		4					4	0.06	subrecedent
Naucoridae	*Thruselinus scutellaris*	Tsc	2				101	1	104	1.67	recedent
Notonectidae	*Anisops bouvieri*	Abo	165		1		35		201	3.23	subdominant
	Anisops breddini	Abr	9		9				18	0.29	subrecedent
	Anisops tahitiensis	Ata	64						64	1.03	subrecedent
	Anisops sp.	Ani	29						29	0.47	subrecedent
Veliidae	*Microvelia leveillei*	Mle	52		1				53	0.85	subrecedent
	Microvelia sp.	Mic		5					5	0.08	subrecedent
Coleoptera											
Dytiscidae	*Rhantus* sp.	Rha	16	2					18	0.29	subrecedent
	Hyphydrus sp.	Hyh	1				1		2	0.03	subrecedent
	Laccophilus sp.	Lac	3	6	1		14		24	0.39	subrecedent
	Neptosternus sp.	Nep	16		1		15		32	0.51	subrecedent
	Copelatus sp.	Cop		1					1	0.01	subrecedent
	Hydrovatus sp.	Hyv		2	2		12		16	0.26	subrecedent
Hydrophilidae	*Amphiops* sp.	Amp	23		25		31	1	80	1.28	recedent
	Helochares sp.	Hel	20	10	14	3	9	4	60	0.96	subrecedent
	Laccobius sp.	Lab	10		5		1		16	0.26	subrecedent
	Berosus sp.	Ber	18	4	3	5	2		32	0.51	subrecedent
	Hydrophilus sp.	Hyp	99	28	20		7	1	155	2.49	recedent
Noteridae	*Canthydrus* sp.	Can	2		8		4	7	21	0.34	subrecedent
	Hydrocanthus sp.	Hyc					2		2	0.03	subrecedent
Scirtidae	*Hydrocyphon* sp.	Hyd		1	1				2	0.03	subrecedent
Spercheidae	*Spercheus* sp.	Spe	23						23	0.37	subrecedent
Diptera											
Ceratopogonidae	*Leptoconops* sp.	Lep		23	6	6	1		36	0.58	subrecedent
Chironomidae	*Chironomus* sp.	Chi	309	241	139	312	165	753	1919	30.8	dominant
	Clinotanypus sp.	Cli	3	59	11	6	6	60	145	2.33	recedent
Culicidae	*Aedes* sp.	Aed	24		8				32	0.51	subrecedent
	Culex sp.	Cul	55		3	2	5		65	1.04	subrecedent
Stratiomyidae	*Odontomyia* sp.	Odo	4		4		12		20	0.32	subrecedent
Syrphidae	*Eristalis* sp.	Eri	47						47	0.75	subrecedent
Ephemeroptera											
Baetidae	*Baetis* sp.	Bae						2	2	0.03	subrecedent
	Cloeon sp.	Clo	18	19	1	3	4	23	68	1.09	subrecedent
Caenidae	*Caenodes* sp.	Cae	5	36		20	2	21	84	1.35	recedent
Trichoptera											
Hydropsychidae	*Amphipsyche meridiana*	Amh		1		29		755	785	12.6	dominant
Total individual			1465	768	636	522	905	1936	6232	100	

Remark: * RA < 1 = Subrecedent; 1.1-3.1 = Recedent; 3.2-10 = Subdominant; 10.1-31.6 = Dominant; >31.7% = Eudominant (Engelmann, 1978)

Odonata, which consisted of three families, accounted for 12.52% of all aquatic insects, with the Libellulidae and *Cratilla* genera dominating. Anisoptera were plentiful in the majority of the bodies of water examined. This could be attributed to their great dispersal capacity and adaptation to a variety of habitats (Adu et al., 2022). Damselflies' low abundance was most likely due to their restricted dispersal abilities, the undulating environment provided by temporary water bodies and a lack of shadow cover (Nagy et al., 2019). The prevalence of damselflies in temporary environments could be attributed to shade from trees surrounding water bodies as well as aquatic vegetation. This confirms Nagy et al. (2019) findings, which demonstrated that shade and aquatic vegetation may benefit Zygoptera over Anisoptera. The prevalence of Libellulidae (Anisoptera) and Coenagrionidae (Zygoptera) in the current study could be attributed to their shorter life cycles, extensive distribution and tolerance to a variety of habitats (Wijesooriya et al., 2022).

Coleoptera had five recorded families, accounting for 7.76%, with the Hydrophilidae and Dytiscidae families dominating. Aquatic Coleoptera can be found in all kinds of freshwater (Luna-Luna et al., 2022) and they exhibit a variety of feeding patterns represented by distinct families (for example, many Dytiscidae are predators, whereas many Hydrophilidae are algivores and detritivores) (Sheth et al., 2021). Dytiscidae have three larval instars that grow in water; all adults are aquatic but may leave during migration or to overwinter on land (Frelik & Pakulnicka, 2015). Both larvae and adult dytiscids are generalist predators in aquatic environments that feed on various species (Ebner et al., 2021). The dytiscid larvae are true predators, but the adults are partially scavengers and larval prey selection is strongly associated with body size (Frelik & Pakulnicka, 2015). In addition to preying on other invertebrates, giant dytiscid larvae may feed on small vertebrates. For example, rising *Dytiscus* larval populations led to increased predation pressure on tadpoles and fish fry (Liao et al., 2024; Frelik, 2014). Hydrophilidae dominates the diversity and richness of Coleoptera in both permanent and temporary ponds (Torres et al., 2012; Macchia et al., 2015). Lutz & Kehr (2017) considered both families to be typical of temporary environments. According to Lutz & Kehr (2017), the abundance of Hydrophilidae and Dytiscidae in temporary ponds is attributable to their remarkable ability to disperse.

Ephemeroptera was less frequent in the families Baetidae and Caenidae, accounting for 2.47% of the aquatic insect population in drainage ditch environments. The family Baetidae can also be found in temporary habitat pools (Reunura & Prommi, 2020) and it is rather common in all types of freshwater alongside the family Caenidae. They are most diverse in unpolluted running water, particularly in the tropics. Although less diverse in standing waters, including genera such as *Cloeon*, the Baetidae accounts for a significant portion of insect biomass in ponds. Most of the Baetidae species are collector-gatherers that feed mostly on detritus (Gattolliata & Nieto, 2009).

Trichoptera was discovered only in the family Hydropsychidae, which includes a significant number of the species *Amphipsyche meridiana*, a lotic species found in the drainage ditches KUKPS2, KUKPS4 and KUKPS6. *A. meridiana* larvae are characterized as omnivore filterers (Permvarunyoo & Prommi, 2013).

Table 3 displays the species diversity indexes. The highest Shannon diversity (H') index of 2.934 was found in KUKPS1, while the lowest (2.118) was reported in KUKPS4, showing a relatively high diversity of aquatic insects in the drainage ditch, which has a variety of microhabitats. Typically, the Shannon index in actual ecological units ranges between 1.5 and 3.5 (Magurran, 2004). The diversity index value can reflect the level of diversity in drainage ditch ecosystems. A higher H′ value implies that the location has a high level of species diversity. The diversity of insects in aquatic ecosystems tends to increase as nutrient levels rise and these optimal environmental circumstances promote their abundance. Their abundance has been related to good food quality and improved water quality in their habitats (Hepp et al., 2013).

Table 3 Number of individuals, species, evenness, Shannon diversity and Simpson indices of aquatic insects in the Kasetsart University's six drainage ditches from October 2022 to March 2023.

Biotic indices	Sampling site					
	KUKPS1	KUKPS2	KUKPS3	KUKPS4	KUKPS5	KUKPS6
Number of individuals	1465	768	636	522	905	1936
Taxon Richness (S)	39	25	33	14	33	20
Evenness index (E)	0.801	0.705	0.660	0.538	0.777	0.506
Shannon diversity (H')	2.934	2.270	2.308	1.420	2.717	1.516
Simpson dominance (D)	0.916	0.841	0.850	0.604	0.906	0.688

The range of evenness values was 0.506 in KUKPS6 and 0.801 in KUKPS1. In the current study, the evenness value was slightly high at almost all sites, indicating that taxa were distributed rather evenly across the environments. The great species diversity and evenness at almost all sites indicate good water quality (Abhijna et al., 2013). The Simpson index ranged between 0.604 in KUKPS4 and 0.916 in KUKPS1. High diversity indices, such as the Shannon-Wiener and Simpson's indexes, suggest that clean or unpolluted water supports a wider range of species, making them practical for detecting organic contamination (Maneechan & Prommi, 2015). Higher numbers of taxa (family) taken from a habitat indicate a more diverse community that typically lives in a healthier environment. Based on the scores, all drainage ditch locations had a relatively diverse aquatic insect fauna.

A cluster analysis using Bray-Curtis distance (Fig. 3) separated the six sampling locations into two groups. The sample stations KUKPS1, KUKPS3 and KUKPS5 had the same unique aquatic insect composition as the other three stations (KUKPS2, KUKPS4 and KUKPS6). The drainage ditch stations KUKPS1, KUKPS3 and KUKPS5 had high taxa richness (39, 33 and 23 genera) and abundance (636 to 1456). These stations also recorded the highest diversity (2.308 to 2.934), as illustrated by Shannon's diversity index.

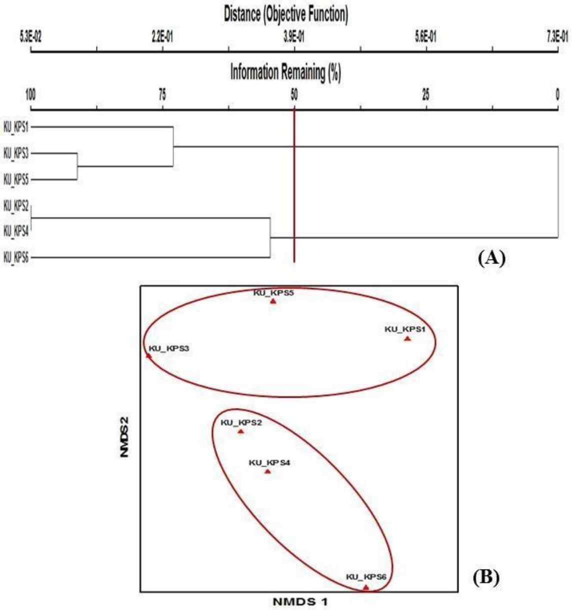

Fig. 3 Cluster analysis (A) and non-metric multidimensional (NMDS) Scaling (B) based on the Bray-Curtis coefficient show similarities in the aquatic insect community at sampling sites.

3. Aquatic insects and environmental parameters

The total variance of the CCA-determined aquatic insect species assemblages was 3.4194. Total variance, or inertia, is the total amount of variability that might be explained. The first three correlations between the biotic and abiotic data sets were 0.970, 0.926 and 0.897, respectively. The first three axes obtained from the CCA explained 38.8% of the variation in aquatic insect species assemblages (Table 4).

The CCA revealed that variables including pH and dissolved oxygen were negatively correlated with the first axis, while water turbidity, nitrate nitrogen and orthophosphate were positively correlated (Table 4). The three first axes of CCA explained 0.588, 0.396 and 0.343 of variation, respectively and were significantly influenced by pH, dissolved oxygen, turbidity, nitrate-nitrogen and orthophosphate (Fig. 4).

Table 4 Canonical Correspondence Analysis (CCA) summary data for aquatic insect and physicochemical water quality variables, such as eigenvalues, variance explained and Pearson correlation values for the first three canonical axes

Number of canonical axes: 3
Total variance ("inertia") in the species data: 3.4194

	Axis 1	Axis 2	Axis 3
Eigenvalue	0.588	0.396	0.343
% variation of species data explained	17.2	11.6	10.0
Cumulative % explained	17.2	28.8	38.8
Species-environment correlation	0.970	0.926	0.897
Water temperature (WT)	-0.260	-0.163	-0.740
pH	-0.596	-0.356	-0.204
DO (Dissolved oxygen)	-0.815	-0.074	-0.210
EC (Electrical conductivity)	0.280	0.334	-0.062
TDS (Total dissolved solids)	0.282	0.335	-0.061
Turbidity	0.602	0.067	0.283
Ammonia-nitrogen (NH_{3}-N)	-0.043	-0.472	0.384
Orthophosphate (PO_4^{3-})	0.575	-0.007	-0.117
Nitrate-nitrogen (NO_{3}-N)	0.547	-0.613	0.184

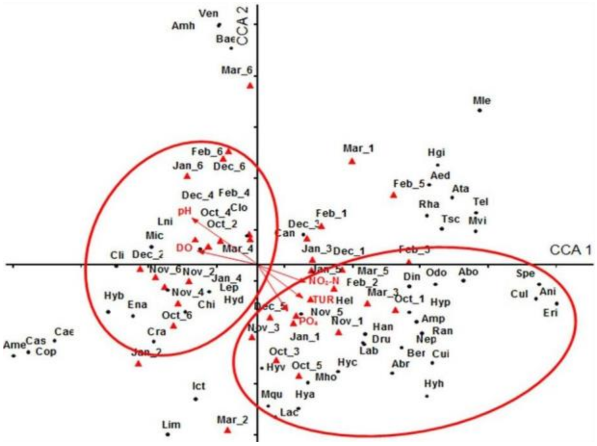

Fig. 4 Presents a triplot of the first and second CCA axes for aquatic insect taxa, environmental factors and sampling locations. Table 2 shows the full names and abbreviation codes for aquatic insect taxa. Monthly codes: Jan = January, Feb = February, Mar = March, Oct = October, Nov = November and Dec = December; drainage ditch Sampling locations: 1, 2, 3, 4, 5, 6.

The eigenvalues for each axis represent the correlation coefficient between sampling sites and environmental variables. An eigenvalue near one indicates a high level of relationship between sampling sites and environmental factors. High eigenvalues (higher than 1) are always linked to long and environmentally solid gradient lines (Palmer, 1993). The present study examined the association between sampling sites and water variables; thus, the total eigenvalue of 1.327 suggested a high degree of correspondence between species and sample sites.

Fig. 4 represents the CCA correlations between aquatic insect data and environmental variables. The CCA ordination diagram clearly separated the sampling site and environmental variables based on aquatic insect species. Each arrow's direction and length represent the variable's maximum rate of change. In the sampling sites KUKPS2, KUKPS4 and KUKPS6, most species depended on dissolved oxygen and water pH. Taxa with negative scores on the first CCA axis included *Cloeon* sp., *Limnogonus* sp., *Clinotarypus* sp., *Cratilla* sp., *Hydrobasileus* sp., *Leptoconops* sp., *Hydrocyphon* sp., *Enallagma* sp., and *Microvelia* sp. Most species at sampling sites KU_KPS1, KU_KPS3 and KU_KPS5 depend on turbidity, nitrate-nitrogen and orthophosphate. Aquatic insect species, such as *Helochares* sp., *Laccophilus* sp., *Hydrotrephes yangae, Hydrophilus* sp., *Amphiops* sp., *Mesovelia horvathai, Diplonychus indicus, Odontomyia* sp., Anisops bouvieri, Spercheus sp., *Culex* sp., and *Eristalis* sp., were among the taxa with positive scores on the first CCA axis.

Conclusion

Based on data on physicochemical water quality and entomofauna composition, the study concluded that Kasetsart University's drainage ditches were slightly polluted. The habitat of the drainage ditch has chironomid assemblages that are resistant to contaminated waters, which provides evidence of this. Furthermore, all of the Odonata taxa collected were eurytopic, meaning they could live in polluted waters. *Cratilla* sp. (Libellulidae), an odonate associated with highly polluted water, was among the Odonata represented at the sampling ditch; however, only a few organisms from this genus were collected. Their incorporation in monitoring, particularly in tropical Asian freshwater locations, will provide additional inferring capability for assessing ecosystem health and contaminated water.

Acknowledgements

The present research was supported by the Faculty of Liberal Arts and Science, Kamphaeng Saen Campus, Kasetsart University. The author would like to thank Dr. Akekawat Vitheepradit and his colleagues for identifying species in the Hemiptera order.

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