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THE RELATIVE IMPACTS OF TOP-DOWN AND BOTTOM-UP PROCESSES ON ZOOPLANKTON BIOMASS AND COMMUNITY BODY SIZE IN URBAN PONDS IN ANKARA

A THESIS SUBMITTED TO
THE GRADUATE SCHOOL OF NATURAL AND APPLIED SCIENCES
OF
MIDDLE EAST TECHNICAL UNIVERSITY

BY

HİLAL KIRAN

IN PARTIAL FULFILLMENT OF THE REQUIREMENTS
FOR
THE DEGREE OF MASTER OF SCIENCE
IN
BIOLOGY

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Approval of the thesis:

THE RELATIVE IMPACTS OF TOP-DOWN AND BOTTOM-UP PROCESSES ON ZOOPLANKTON BIOMASS AND COMMUNITY BODY SIZE IN URBAN PONDS IN ANKARA

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ABSTRACT

THE RELATIVE IMPACTS OF TOP-DOWN AND BOTTOM-UP PROCESSES ON ZOOPLANKTON BIOMASS AND COMMUNITY BODY SIZE IN URBAN PONDS IN ANKARA

Kıran, Hilal
Master of Science, Biology
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Ponds are small and shallow water bodies that are rich in biodiversity as they provide different habitats, food, and water to aquatic and many terrestrial species. One of the biggest threats to biodiversity is urbanization because it causes increased land use, habitat fragmentation, and removal of riparian vegetation. Increased land use affects dramatically the urban pond ecosystem, i.e, changing the food web structure that is critical for the energy transfer between different trophic levels. Since zooplankton connect phytoplankton producers to higher trophic levels such as fish, changes in resource availability and predation pressure as a result of urbanization can have a significant effect on their biomass and size structure. This study aimed to assess the relative impacts of top-down and bottom-up processes on zooplankton biomass and abundance-weighted mean community body size in urban ponds in Ankara. The first hypothesis was that if bottom-up processes are dominant, zooplankton biomass and size will be associated with the phytoplankton abundance, as well as abiotic variables that influence phytoplankton density such as water clarity and nutrient concentrations.

The second hypothesis was that if top-down processes are dominant, zooplankton biomass and size will be associated with predator presence and abundance. There was a positive relationship between total nitrogen (TN) concentrations with biomass and the size of total zooplankton and copepods, indicating the nitrogen limitation of the ponds, and the possible importance of the benthic zone in the trophic structure. Additionally, macroinvertebrates showed a positive relationship with total zooplankton size, and copepods' biomass and size, indicating possible mouth-gape-limited predation on smaller zooplankton. To sum up, these findings suggest that zooplankton biomass and size in urban ponds in Ankara are influenced by both top-down (macroinvertebrate predation) and bottom-up (nutrient availability) forces.

Keywords: Urbanization, Trophic Cascade, Pond, Zooplankton

ÖZ

ANKARA’DA KENTSEL GÖLCÜKLERDE YUKARIDAN AŞAĞIYA VE AŞAĞIDAN YUKARIYA SÜREÇLERİN ZOOPLANKTON BİYOKÜTLE VE KOMÜNİTE VÜCUT BÜYÜKLÜĞÜ ÜZERİNDEKİ GÖRELİ ETKİSİ

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Gölcükler, birçok türe farklı yaşam alanları, yiyecek ve su sağladıkları için biyolojik çeşitlilik açısından zengin olan küçük ve sığ su kütleleridir. Biyoçeşitliliğe yönelik en büyük tehditlerden biri kentleşmedir çünkü artan arazi kullanımına, habitat parçalanmasına ve nehir kıyısındaki bitki örtüsünün kaldırılmasına neden olmaktadır. Artan arazi kullanımı, kentsel gölcük ekosistemini önemli ölçüde etkiler, yani farklı trofik seviyeler arasındaki enerji transferi için kritik olan besin ağı yapısını değiştirir. Zooplankton, üreticiden tüketiciye enerji akışının ortasında olduğundan, kaynak mevcudiyeti ve avlanma baskısı gibi çevredeki herhangi bir değişiklik, biyokütle ve vücut büyüklükleri üzerinde önemli bir etkiye sahip olabilir. Bu çalışmanın amacı, yukarıdan aşağıya ve aşağıdan yukarıya süreçlerin zooplankton biyokütlesi ve bolluk-ağırlıklı ortalama komünite vücut büyüklüğü üzerindeki göreli etkisini Ankara'nın kentsel gölcüklerinde belirlemekti. İlk hipotez, aşağıdan yukarıya süreçlerin baskın olması durumunda, zooplankton biyokütlesi ve vücut boyunun, fitoplankton yoğunluğu ve besin konsantrasyonundaki değişikliklerle ilişkili olacağı, ayrıca; su berraklığının fitoplankton yoğunluğuyla birlikte zooplankton biyokütlesi ve vücut büyüklükleri üzerinde etkisi olacaktır.

İkinci hipotez, yukarıdan aşağıya süreçler baskınsa, zooplankton biyokütlesi ve vücut büyüklüklerinin avcı varlığı ve bolluğu tarafından kontrol edilebileceğiydi. Toplam nitrojen (TN) konsantrasyonlarının, toplam zooplankton ve kopepodların biyokütleleri ve vücut büyüklükleri ile pozitif bir ilişki gösterdiği, bu da gölcüklerin nitrojen sınırlamasına ve bentik bölgenin trofik yapıdaki olası önemine işaret ettiği bulunmuştur. Ek olarak, makroomurgasızlar, toplam vücut büyüklükleriyle, ve kopepodların biyokütleleri ve vücut büyüklükleriyle pozitif bir ilişki gösterdi ve bu, daha küçük zooplankton üzerinde olası avlanmaya işaret ediyor. Özetlemek gerekirse, bu bulgular iki hipotezi desteklemektedir; Ankara'nın kentsel gölcüklerindeki zooplanktonların komünite yapısı besin mevcudiyeti ve makroomurgasızların predasyonuyla önemli ölçüde etkilemektedir.

Anahtar Kelimeler: Kentleşme, Trofik Seviye, Gölcük, Zooplankton

To Donald…

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INTRODUCTION

Ponds are small and shallow waterbodies that differ from lakes and wetlands regarding water chemistry, area, and depth. Ponds are very rich in biodiversity, providing different habitats, food, and water to aquatic and many terrestrial species, including birds and insects (Nummi et al. 2011). Even though ponds are smaller and shallower, they are one of the main hotspots for biodiversity; therefore, they are critical for biodiversity conservation (Biggs et al.,1994). Ponds also have a critical role in providing ecosystem services to society such as water purification, carbon sequestration, habitat for endemic species, climate regulation, and flood control (Jiang et al., 2011; Maltby and Acreman, 2011). The ecosystem services that ponds provide depend on the pond's physical and chemical characteristics and the surrounding area. While ponds form naturally, they can also be human-made, primarily for agricultural and aesthetic reasons.

Urbanization is the biggest threat to biodiversity because it changes land use, causes habitat fragmentation, changes dispersal patterns, and causes the removal of riparian vegetation from water bodies. It is estimated that 55% of the world’s population now lives in urban areas, and this is expected to increase to 68% by 2050 (United Nations, 2018). Urban ponds show different environmental properties to nonurban ponds; they are under different stressors and mainly have concrete margins, reduced vegetation cover, lower connectivity to other waterbodies, and are exposed to runoff from residential and industrial developments that can cause eutrophication (Hassall, 2014).

One of the most damaging stressors is increasing land use. It may cause an increase in the disposal of phosphorous and nitrogen into aquatic ecosystems, and excess nutrient accumulation can give rise to eutrophication which results in poor water quality, reduced biodiversity, brownification, harmful algal blooms (HABs), and fish kills (Brönmark and Hansson, 2002, Grimm et al., 2008). Furthermore, brownification is caused by increased runoff of dissolved organic carbon (DOC) from the pond surrounding and is important for water quality as well as the structure of the aquatic ecosystems (Solomon et al., 2015). Browning affects productivity with a unimodal relationship. Until a certain point, DOC increase favors the biological productivity in lakes; however, when it exceeds 5 mg/L concentration, it decreases productivity (Seekell et al., 2015). Also, browning affects food webs dynamics by changing the water clarity, affecting prey and visual-predator interaction. Start et al. (2019) showed that urbanization affects species and community-level patterns of diversity and that also causes changes in the food webs structure.

Food web structure is critical for how energy flows across an ecosystem, but it is difficult and complex to understand. An organism's trophic level depends on what organisms eat (King, 2019). Trophic cascade theory states that resource availability at the top of the food web shapes the abundance, biomass, and productivity of primary producers (Carpenter & Kitchell, 1996). It contains two types of processes: bottom-up and top-down. A bottom-up process influences the abundance of organisms on higher trophic levels through changes in resource availability while a top-down process influences the abundance of organisms on lower trophic levels through predator-prey interactions (Hairston et al., 1960; Polis et al., 2000).

For example, a typical trophic cascade in the freshwater ecosystem is indicated in Figure 1.1 (Carpenter & Kitchell, 1996). Top predators such as fish may feed on macroinvertebrates, zooplankton, or other smaller fishes. Macroinvertebrates may also feed on algae, zooplankton, or other smaller macroinvertebrates. Moreover, zooplankton may feed on phytoplankton, periphyton, and other smaller zooplankton (Hoar & Randall, 1969).

Zooplankton are one of the most critical organisms in freshwater ecosystems since they are in the middle of energy flow from producer to consumer. They can affect water quality, algae densities, fish production, and nutrients (Carpenter & Kitchell, 1996).

They are also sensitive to environmental changes such as a change in temperature, salinity, etc (Brucet et al., 2010; Akbulut & Tavşanoğlu, 2018). Most studies show that zooplankton biomass and body size are mainly shaped by the quantity and quality of their food resources (Persson et al., 2007; Müller-Navarra,2008; Brett et al., 2009), and predation pressure (Hessen et al., 1995).

Resource availability represents bottom-up processes that shape the zooplankton community. Phytoplankton is their main food source, which needs nutrients such as phosphorous (P) and nitrogen (N) for its growth (Conley et al., 2009). Therefore, nutrient levels have a critical indirect role in zooplankton biomass and size structure. Many studies showed that increased nutrient favors phytoplankton growth (Shurin et al., 2012). For example, studies are showing that nutrient-enriched lakes lead to an increase in zooplankton biomass and density through higher food availability (Pinto- Coelho et al. 2005). On the other hand, nutrient enrichment can cause eutrophication, leading to harmful algal blooms as well as increased planktivorous fish predation that can also affect primary productivity (Jeppesen et al., 2003). There are some laboratory experiments with Daphnia as a model organism to show algal blooms- zooplankton interactions, resulting in a reduction in the number of individuals with a body length of >1 mm when cyanobacteria reached high biomass (Ghadouani et al., 2003).

Water clarity also has a crucial role in shaping the zooplankton community since primary productivity depends on it. Reduced water clarity i.e., shading reduces primary productivity (Thrane et al., 2014) that results in reduced food availability for zooplankton.

On the other hand, predation pressure represents top-down processes that shape the zooplankton community. Predation by fish and macroinvertebrates causes changes in the zooplankton community.

Many studies show fish predation reduces zooplankton biomass (Christoffersen et al., 1993; Jeppesen et al., 1997), and when planktivorous fish biomass increases, they will mostly feed on larger zooplankton, and predation mainly reduces the mean body size of the zooplankton (Brooks and Dodson 1965; Lemmens et al. 2018), so smaller zooplankton will dominate the community (Williams & Moss, 2003). Another example also shows that planktivorous fish predation causes a decrease in big cladocerans (e.g Daphnia;) abundance and favors small cladocerans, copepods, and rotifers (Beklioğlu & Moss, 1996).

Moreover, water clarity has an indirect role in shaping the zooplankton community through predators. Elevated levels of dissolved organic carbon (DOC) can attenuate light, and reduced light intensities interfere with visual predators. Many studies showed that reduced light intensity decreases the reactive distance of planktivorous fish (Vinyard and O'Brien 1976), and their predation rate (Persson 1986; Bergman 1987). Also, other studies showed that planktivorous fish select larger prey at sufficient light levels such as macroinvertebrates (Brooks and Dodson,1965; Taylor 1980). This increases zooplankton biomass by changing prey organisms and favors larger body-sized zooplankton from visual predation.

Macroinvertebrates are also important predators of zooplankton, especially in fishless ponds. For example, Notonectids prey on *Daphnia*, resulting in reduced biomass (Arnér et al., 1998). On the other hand, indirect interactions between different macroinvertebrate groups may favor the zooplankton community. Cobbaert et al. (2010) showed that top-down effects of predatory *Dytiscus alaskanus* (Coleoptera: Dytiscidae) on other predatory invertebrates led to an increase in total zooplankton biomass due to increased abundance of large and small cladocerans.

Moreover, some macroinvertebrates are mouth-gape limited such as Chaoborus, which are limited to smaller-bodied zooplankton and can shift community size distribution to larger individuals (Hall et al., 1976; Zaret, 1980).

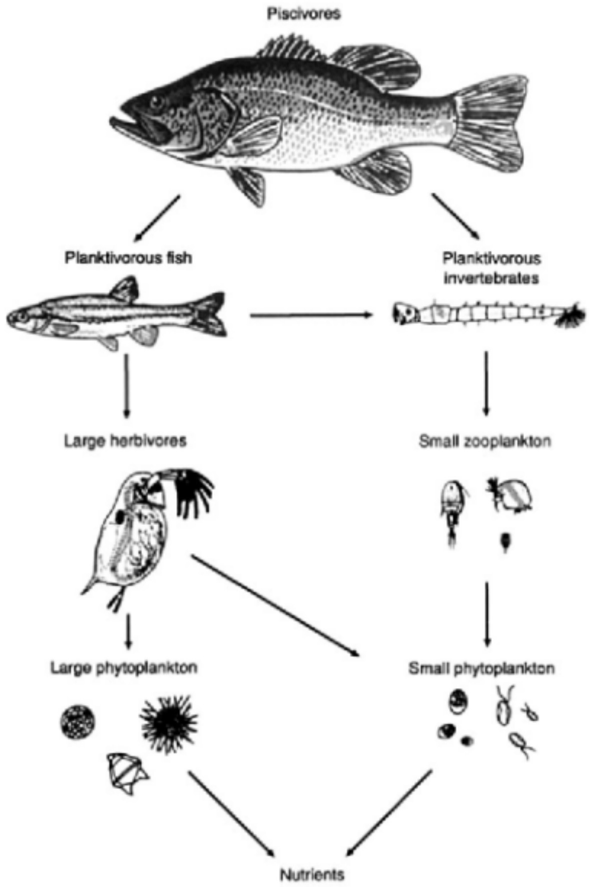

Figure 1.1. Major interactions of the trophic cascade in experimental lakes. The top predator is piscivores fish, feeding on planktivorous fish. Planktivorous fish feed on zooplankton and invertebrates, planktivorous invertebrates feed on zooplankton, and zooplankton feeds on phytoplankton. Adapted *from The Trophic Cascade in Lakes* (1^{st} ed., p. 5) by 1996, Cambridge University Press.

The goal of this study was to determine the relative impacts of top-down and bottom-up processes on zooplankton biomass and community body size in urban ponds in Ankara, Turkey. The first hypothesis was that if bottom-up processes are dominant, zooplankton biomass and size will be associated with the changes in phytoplankton density and nutrient concentration. In addition, there will be an interactive effect of water clarity and phytoplankton density on zooplankton biomass and size.

The first prediction was that nutrient (total phosphorous, total nitrogen) availability may increase the productivity of the ponds; this indirectly affects the biomass and size of zooplankton positively as it increases primary production. Many studies showed that increased nutrients promote both pelagic and benthic primary producers' growth (Shurin et al.,2012). The second prediction was that water color or brownification of water (measured as dissolved organic carbon) will limit light penetration in the ponds, which will indirectly reduce the food availability for zooplankton, which might indirectly affect zooplankton biomass and size negatively.

The second hypothesis was "If top-down processes are dominant processes, zooplankton biomass and size may be controlled with predator presence and abundance.". The third prediction was that when zooplanktivorous fish are present and abundant in the ponds because they are visual predators, it will cause predation pressure on larger community body-sized zooplankton. The community will be driven to be dominated by smaller community body-sized zooplankton and caused reduced biomass. The fourth prediction was that when macroinvertebrates are abundant in the ponds, it will cause predation pressure on smaller community body-sized zooplankton because of mouth-gape limitation. Therefore, the community will be driven to be dominated by larger community body-sized zooplankton and cause increased biomass.

CHAPTER 2

MATERIALS AND METHODS

This study was performed within the framework Ponderful project (POND Ecosystems for Resilient Future Landscapes in a changing climate), an H2020 "Research and Innovation Programme" project funded by the European Union. The main goal is to develop improved methods for maximizing the use of ponds and pondscapes to mitigate and adapt to climate change, protect biodiversity, and deliver ecosystem services (Ponderful, 2020). The sampling protocol that was created by this project was applied in this study.

2.1 Study Area and Sampling

This study was performed in 17 ponds located in 3 pondscapes, which were located in the southern part of Ankara (39°53'50.98"N; 32°54'10.27"E- 39°44'14.22"N; 32°46'35.72"E), starting from Imrahor River Valley region through the South of Lake Mogan (Figure 2.1). Sampling started on the 3rd of June 2021, and ended on the 5th of September 2021. Thus, the pond codes of the studied ponds were UP1, UP2, UP4, UP5, UP6, UP7, UP9, UP, UP10, UP11, UP12, UP13, UP14, UP16, UP17, UP18, UP19, and UP20 *(UP stands for urban ponds*).

Imrahor River Valley pondscape (Figure 2.1a) area is approximately 2.41 km^2. This pondcscape is under the threat of urbanization and domestic pollution. Animal farming is also common in this area; this causes disturbed sediment because cattle entry to the ponds and cattle dung accumulates in and around the pond (personal observation).

Gölbaşı Düzlüğü pondscape (Figure 2.1b) area is approximately 0.26 km^2. It is part of the Gölbaşı Special Environmental Protection Area and declared an “Important Bird Breeding & Shelter Area”. This pondscape is located downstream of Lake Mogan, it is anticipated that the pondscape is like a buffer zone against hydro-meteorologic hazards such as flooding (Z. Akyürek communication).

Lake Mogan Pondscape (Figure 2.1c) area is approximately 0.58 km^2. It is part of the Gölbaşı Special Environmental Protection Area and declared an “Important Bird Breeding & Shelter Area”. This pondscape is also under a high urbanization risk. Around Lake Mogan Pondscape, 102 bird species are observed, including the endangered species white-headed duck (*Oxyura leucocepahala*).

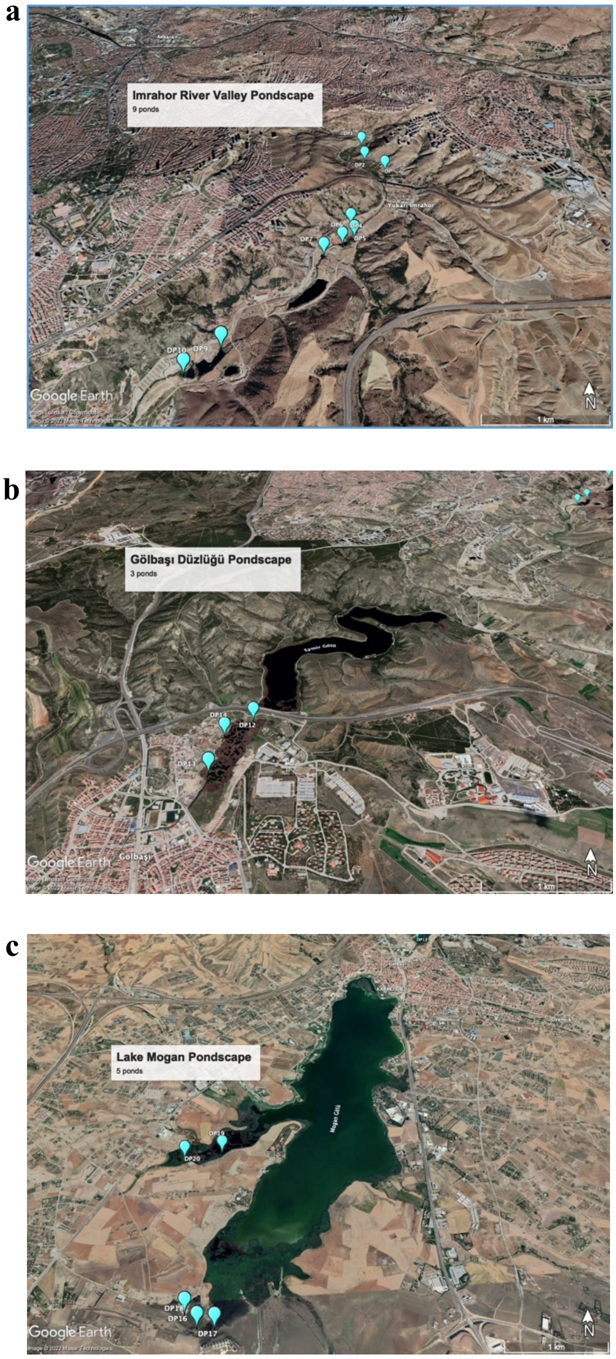

Figure 2.1. Maps of urban ponds sampled in Ankara. All ponds lie within the coordinates 39°53'50.98"N and 32°54'10.27"E to 39°44'14.22"N and 32°46'35.72"E. (**a**) *Imrahor River Valley Pondscape*. (**b**) *Gölbaşı Düzlüğü Pondscape* (**c**) *Lake Mogan Pondscape*

2.2 Physical and Chemical Variables

Conductivity, pH, oxygen concentration, and water temperature were measured at the same point every 0.5m using a multiprobe YSL. Secchi disc was also measured. The depths of the ponds were measured by a depth meter. Depth-integrated water sampling covering the entire water column was done at the deepest point of the pond, in every 0.5 m, water was collected with a Ruttner sampler. Water samples were poured over a 250 µm mesh to remove large material, and samples were used for TP (Total phosphorous), TN (Total Nitrogen), DOC (Dissolved Organic Carbon), SS (suspended solid), and chlorophyll-a (Chl *a*) analyses. When there was a thermal stratification, water samples were collected separately from both layers. Then, the samples were frozen at -20 C until the analysis.

2.2.1 Water Chemistry Analysis

Total phosphorus (TP) was determined by following the molybdenum blue method described by Machereth et al., 1978. Total nitrogen (TN) determination, including ammonium (NH_4^+), and nitrate (NO_3) were performed by using an automated wet chemistry analyzer (Baird & Bridgewater, 2017). For the determination of suspended solid (SS), water was filtered from pre-weighted GF/C Glass microfiber filters (1.2 µm pore size, Whatman International), then put into the oven for drying the filters, and re-weighting the samples' total dissolved solids were calculated (APHA, 1926). Chl *a* analysis was carried out by filtering 0.5 L of water from each pond through GF/C Glass microfiber filters (1.2 µm pore size, Whatman International), then extracting chlorophyll-*a* with ethanol and reading the absorbances at 663 nm and 750 nm in a quartz cuvette (Jespersen et al., 1987) by using a spectrophotometer (NanoPhotometerTM P-360). TOC and DOC analysis was performed in a private laboratory following the TS 8195 EN 1484 standardization methodology (TSE, 2000).

2.3 Biological Sample Collection

Ponderful protocol (2020) was followed for the collection, preservation, and counting of the zooplankton and macroinvertebrates.

2.3.1 Zooplankton Sample Collection, Identification, and Counting

Zooplankton samples were collected with a tube sampler from 8 different points. A predefined grid was used to decide those 8 locations to ensure that different sub-habitats of the pond area are represented in the sample. Six liters of the whole water column were collected in every 8 points. The water collected from the entire pond (48L) was collected in a large bucket. After stirring eight times, 40L of the water was filtered through a 53 µm conical plankton net and collected in a 100ml amber bottle. The samples were preserved in 4% Lugol solution (Sigma Aldrich). A month after sampling, a smell from the samples was detected. To preserve the samples better, a 4% final concentration of formaldehyde saturated with glucose (a small spoon of table sugar) was added. After the formaldehyde addition, no more smell was detected. The zooplankton samples were counted with Leica DFC295 digital microscope and the LAS V4.12 software.

For counting, 8-10 ml of sub-samples were taken from each bottle using a Pasteur pipette, and it is diluted with distilled water in the counting plate to ease counting. Subsampling and counting continued until 300 individuals from each species were counted. Moreover, the length of the first encountered 25 individuals from each species was measured. For copepods, measurement was performed from the anterior tip to the end of the caudal ramus. For cladocera, they were measured from the top of the head to the base of the tail spine. Nauplii and copepodites were counted; however, their body sizes were not measured. For taxonomical identification, the keys developed by Harding & Smith, 1974, and Scourfield & Harding, 1966 were used. Also, the website "An-Image-based Key to the Zooplankton of North America" was used.

From each species, 25 individuals' body sizes were measured whenever possible. When there were not 25 individuals, the maximum number of species was measured. Abundance-weighted mean community body size for each species was calculated in each urban pond using the formula described in research by Brans et al (2017) using the following formula;

$$\bar{z}^{j}_{\mathrm{L}} = \sum_i q_{ij} z_{ij}$$

In which, for pond j, q_{ij} was relative abundance, z_{ij} was the average body size value of species i of pond j. In details, the mean body size of 25 individuals for each species were calculated, then this mean value was multiplied by relative abundance of that species to assign one body size value for each species with respect to its abundance in this community. By summing up each species' abundance-weighted mean body size from the same group, different abundance-weighted mean community body sizes were calculated for copepods and cladocerans. To calculate the total crustacean zooplankton (cladocerans and copepods), two groups were calculated separately using this formula and summed together.

Biomass of total crustacean zooplankton and biomass of separate zooplankton groups were calculated by using standard allometric equations to convert body lengths to biomass (Dumont et al., 1975; McCauley,1984) since this study mainly focused on community structure.

Naupli and Copepodid were not included in the biomass and size calculations since only adults of the species' body length were measured.

2.3.2 Macroinvertebrates Sample Collection, Identification, and Enumeration

The samples were taken with a sweep-net with a mesh size of 500 µm, and with a 25*15 cm frame size, see also Figure 2.2. Sampling was performed mostly in the littoral vegetation and the open water area where the submerged macrophytes and the floating leaved macrophytes were dominant. 20 sweeps of 1 m in each pond were sampled in each mesohabitat. Sampling was conducted by walking around the pond, then the net was pooled into a big tray and was washed carefully to take all material. The large invertebrates were taken into 250 ml bottles with %70 ethanol in the field, and the remaining sample was brought to the laboratory in a 3L jar with 70% ethanol. Then, invertebrates from the jar were cleaned from the debris under a stereomicroscope (Leica M125, Wetzlar, Germany) under 10x magnification and put into 250 ml bottles with %70 ethanol for further identification.

Figure 2.2. Macroinvertebrate sampling net with 25x15 cm with mesh size 500 µm, and 1.5 m long.

For taxonomical identification of macroinvertebrates, the samples were put into a petri dish and diluted with distilled water. Utility forceps, stainless steel with curved pointed ends, were used to turn the samples.

A stereomicroscope (Leica M125, Wetzlar, Germany) was used under 10x magnification. Specimens belonging to the Odonata, Coleoptera, Trichopteran, Gastropod, Ephemeroptera, Diptera, Hemiptera, Isopod, and Prostigmata were sorted and identified at the lowest taxonomic level possible, mainly family. The keys developed by MacAn (1972), Gooderham (2002), Thyssen (2009), Oscoz et al. (2014), and Allan et al. (2021) were used for identification. Then, zooplanktivorous macroinvertebrates belonging to the families *Dytiscidae* (Water tigers), *Culicidae* (Mosquitoes), *Notonectidae* (Backswimmers), *Coenagrionidae, and Libellulidae* were enumerated. Ponds were categorized as low and high macroinvertebrates concerning the density of macroinvertebrates.

2.3.3 Fish Sample Collection, Identification, and Counting

Sampling was performed in late September and continued for two weeks. Fish nets with a length of 30 m, a height of 1.5 m, and 12 different mesh apertures ranging from 5 mm to 55 mm were used. Nets were put in concerning pond areas; when it is larger than 1 hectare, 2 nets were put. Nets were put in the mornings and collected 3 hours later. The nets were hung on stakes; fish were collected and identified at the species level. Nets were not put in three ponds that had dried out UP1, UP17, and UP20.

For littoral nets, the nets were put along the macrophytes belt, and the pelagic nets were parallel to the littoral zones. For the irregular pond's shape, the direction was arranged accordingly. For taxonomical identification, the keys developed by McPhail and Carveth (1993), Maitland & Linsel (2006) were used. Catch per unit effort (CPUE) was used as an index of fish abundance, it was calculated using the following formula by Hubert & Fabrizia (2007):

$U = \frac{\sum C}{\sum f}$ In which, U was catch per unit effort, C was catch and f was an effort. C was measured as the number of fish caught, and f was measured as (nets x hours).

2.4 Statistical Analysis

All analyses were conducted in RStudio (release 2022-07-1) with R version (R 4.0.3 GUI 1.73 Catalina build (7892)). "Spotted Wakerobin" Release using packages, robustbase version 0.95-0, piecewiseSEM v 2.1.1 with $\alpha = 0.05$. For the visualization, ggplot2 v 3.3.6 was used.

For the multivariate analysis, piecewise structural equation modeling (piecewise SEM, Shipley 2000, Grace 2006) was used in the software R using the packages ‘dagitty,’ and ‘piecewiseSEM’ (Lefcheck, 2016) to assess the relative impacts of variables associated with top-down (macroinvertebrates and fish) and bottom-up (total phosphorous, total nitrogen, dissolved organic carbon, and Chlorophyll a) processes on abundance-weighted mean community body size and zooplankton biomass. Since the sample size was low, traditional regression techniques could not be utilized to assess all possible relationships. A p-value less than 0.1 was considered as a significant interaction. A separate piecewise SEM model was created for copepod and cladoceran abundance weighted mean body size and biomass. Significant paths for the model were fit with GLMs with a normal distribution and a log link function. All GLMs, were fit either log-normal or gamma distribution based on Akaike’s Information Criterion (AIC), with the model with the lowest AIC score chosen for the model selection. Log-likelihood ratio tests were used to identify the minimum adequate model following Crawley's (2005) procedure using the Chi-square test. Model fit was also visually assessed using, plots of residuals versus fitted values and the square root of the standard deviance of residuals versus fitted values. Furthermore, Cook’s distance was used to detect out influential points **(outliers)** with leverage greater than 1.0. When influential points were identified, robust GLMs were used fitting a Gamma or log-normal distribution with Mallows or Huber-type robust estimators (Cantoni and Ronchetti ,2001; Cantoni and Ronchetti, 2006) that down weights the effect of influential points on model fit.

CHAPTER 3

RESULT

3.1 Description of the ponds

All sampled ponds were in similar elevations, and the areas ranged from 0.02-4.43 hectares with a maximum depth in the range of 0.15 to 6.0 meters (Table 3.1). One of the sampling ponds was shown in Figure 3.1. Five of the ponds had an area equal to or larger than 1 hectare, and two had a maximum depth equal to or larger than 5 meters.

Figure 3.1. One of the sampling ponds in Imrahor Valley Pondscape during the sampling (*Photo is taken in May 2021).*

Secchi disc transparency changed greatly among ponds starting from 15 cm to 115 cm *(see S2 in supplementary*). The lowest water temperature value was 14.04 °C, while the highest was 25.96 °C. Seven of the ponds had a temperature equal to or greater than 20 °C, while the rest had a temperature lower than 20°C.

Table 3.1 Physical characteristics of 17 urban ponds (UP), PS=Pondscape, IM= Imrahor River Valley pondscape, GD=Gölbaşı Düzlüğü pondscape, MO=Lake Mogan pondscape, area (ha), Maximum depth(m), Secchi depth(cm), and temperature (°C).

PS	***Pond Code***	***Area (ha)***	***Max. Depth (m)***	***Secchi Depth (cm)***	***Temperature (°C)***
IM	UP1	0.02	0.15	15	25.34
IM	UP2	0.03	2.5	55	16.78
IM	UP4	0.33	3.5	99	19.02
IM	UP5	0.22	2.0	105	17.76
IM	UP6	0.48	4.0	76	16.13
IM	UP7	0.23	4.2	77	15.22
IM	UP9	2.54	6.0	54	14.04
IM	UP10	0.98	3.8	63	14.45
IM	UP	0.14	2.8	115	14.59
GD	UP12	0.32	2.2	98	16.49
GD	UP13	1.0	3.0	47	20.51
GD	UP14	0.44	5.9	87	15.84
MO	UP16	1.14	0.7	50	22.16
MO	UP17	0.54	0.26	26	25.96
MO	UP18	0.46	0.6	39	22.97
MO	UP19	4.43	1.3	76	24.17
MO	UP20	1.29	0.5	21	22.11

Chemical variables are shown in Table 3.2. Dissolved Oxygen (DO) values varied among ponds. The highest value was 11.8 mg/L, which was in Imrahor River Valley pondscape. The lowest value was 0.3 mg/L, which was in Gölbaşı Düzlüğü pondscape.

Suspended solid (SS) values ranged from 9.1 mg/L to 143.5 mg/L, with the lowest value observed in Lake Mogan pondscape, while the highest value was in Gölbaşı Düzlüğü pondscape.

Dissolved organic carbon (DOC) concentration ranged from 4 mg/L to 79.8 mg/L (see Table 3.2). The lowest concentration was observed in Imrahor River Valley pondscape, while the highest value was in Lake Mogan pondscape. On the other hand, when comparing the pondscapes with respect to the average values (see Figure 3.2), the highest average DOC concentration was observed in Imrahor River Valley pondscape, and the lowest in Gölbaşı Düzlüğü pondscape.

In all ponds total phosphorous (TP) concentration was higher than 100 µg/L. The highest TP concentration was 491.4 µg/L, while the lowest was 121.2 µg/L; both were recorded in Imrahor River Valley pondscape. However, when comparing the pondscapes concerning the average concentration (see Figure 3.2), the highest average concentration was observed in Gölbaşı Düzlüğü pondscape, and the lowest in Lake Mogan pondscape. Furthermore, the soluble reactive phosphorous (SRP) concentration ranged from 64.6 to 504.9 µg/L in Imrahor River Valley pondscape.

In all ponds, the total nitrogen (TN) concentration was above 1000 µg/L. The highest TN concentration was 6035.1 µg/L which was in Imrahor River Valley pondscape. However, when comparing the pondscapes with respect to the average concentrations (see Figure 3.2), the highest average TN concentration was observed in Gölbaşı Düzlüğü pondscape, and the lowest in Imrahor River Valley pondscape. Dissolved inorganic nitrogen (DIN) concentrations ranged from 30.2 µg/L to 1694.7 µg/L, the highest DIN concentration was observed in Gölbaşı Düzlüğü pondscape while the lowest DIN concentration in Imrahor River Valley pondscape.

TN/TP mass ratio values (Figure 3.3) were lower than 10 in 1/3 of the ponds. TN/TP mass ratio values were lower than 20 in most ponds (82.3%), and TN/TP mass ratio values were higher than 20 in> 17.6% of the ponds. One pond from Imrahor River Valley pondscape, one pond from Gölbaşı Düzlüğü pondscape, and two ponds from Lake Mogan pondscape had rather higher TN/TP mass ratio values.

Chlorophyll-*a* (Chl *a*) concentration differed greatly; the lowest value was 2.3 µg/L, while the highest concentration was up to 525.6 µg/L, both were in Imrahor River Valley pondscape. However, when comparing the pondscapes concerning the average concentrations (see Figure 3.2), the highest average Chl *a* was observed in Gölbaşı Düzlüğü pondscape, and the lowest in Lake Mogan pondscape.

Table 3.2 Chemical characteristics of urban ponds (UP). PS=Pondscape, IM= Imrahor River Valley pondscape, GD=Gölbaşı Düzlüğü pondscape, MO=Lake Mogan pondscape, DO=dissolved oxygen, SS=suspended solid, DOC=dissolved organic carbon, TP=total phosphorous, SRP=soluble reactive phosphorous, TN=total nitrogen, DIN=dissolved inorganic nitrogen, TN/TP =TN/TP mass ratio, Chl *a*= phytoplankton abundance.

PS	***Pond Code***	***DO (mg/l)***	***SS (mg/l)***	***DOC (mg/l)***	***TP (µg/l)***	***SRP (µg/l)***	***TN (µg/l)***	***DIN (µg/l)***	***TN/ TP***	***Chl a (µg/l)***
IM	UP1	11.8	21.7	62.8	121.2	64.6	2862.3	65.0	23.6	2.3
IM	UP2	0.4	17.6	16.0	394.9	288.2	4760.6	1299.9	12.0	525.6
IM	UP4	2.2	18.5	13.76	383.9	309.0	2991.8	628.2	7.8	112.4
IM	UP5	5.6	21.3	4.52	386.5	131.6	1006.6	30.2	2.6	52.8
IM	UP6	3.9	35.4	4.00	491.4	187.6	2194.7	849.8	4.5	28.2
IM	UP7	6.9	25.3	7.28	378.4	282.9	2133.7	555.1	5.6	131.7
IM	UP9	3.6	19.8	14.20	444.8	494.0	6035.1	1132.4	13.6	91.2
IM	UP10	6.2	28.9	21.75	402.4	504.9	5444.3	1445.0	13.5	176.6
IM	UP	0.8	130.3	6.37	296.4	163.7	3507.8	1379.9	11.8	115.4
GD	UP12	0.3	21.0	7.08	420.0	337.4	5629.9	1694.7	13.4	361.6
GD	UP13	2.3	143.5	22.25	337.9	191.7	4813.8	1097.6	14.3	124.8
GD	UP14	1.6	22.6	21.05	440.4	225.9	4092.81	1000.2	9.3	132.3
MO	UP16	1.8	38.7	71.10	130.9	96.4	4081.52	93.8	31.2	26.9
MO	UP17	5.1	16.5	72.60	257.0	111.0	4779.88	60.5	18.6	8.9
MO	UP18	1.9	28.0	79.80	424.28	78.0	4152.12	50.8	9.8	31.8
MO	UP19	0.6	9.1	53.50	156.41	20.3	3483.88	124.5	22.3	9.3
MO	UP20	1.30	119.0	21.00	385.22	167.5	4828.36	202.5	12.5	67.1

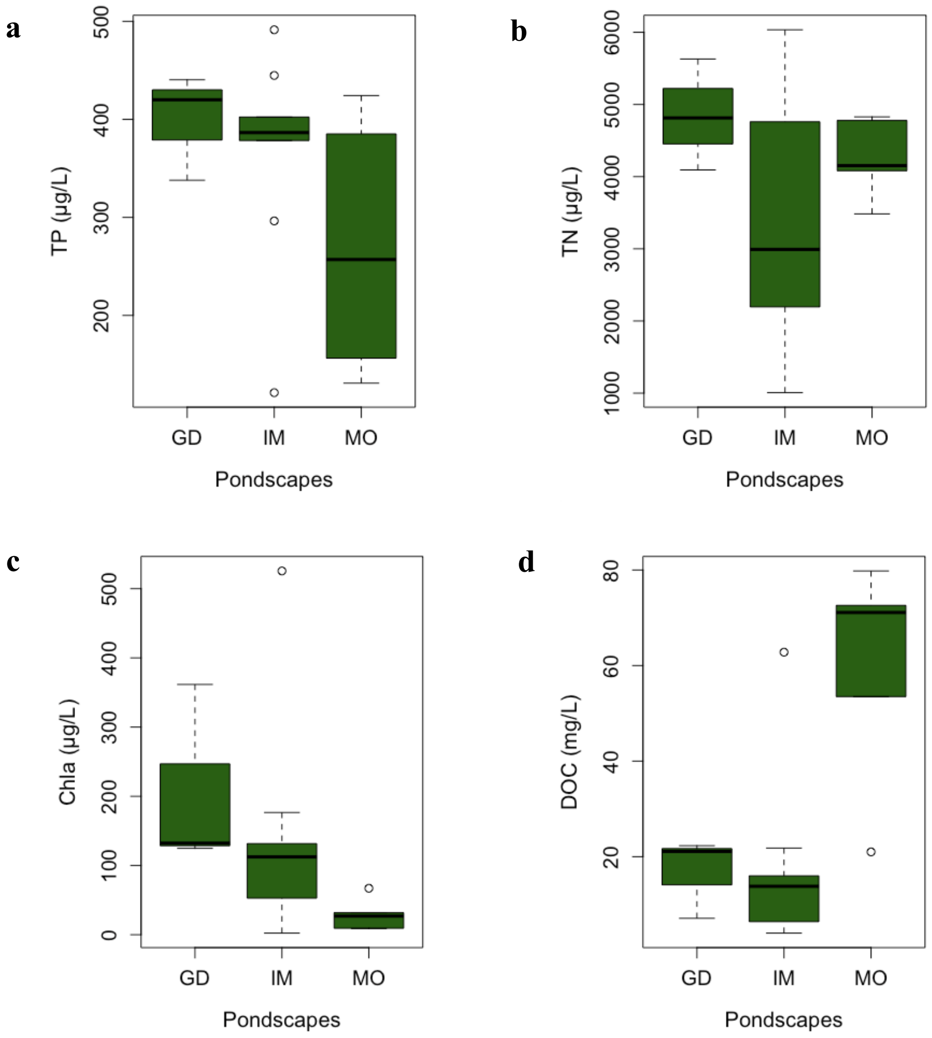

Figure 3.2. The mean of chemical characteristics of urban ponds (UP) for each pondscapes; Gölbaşı Düzlüğü Pondscape(GD), Imrahor River valley Pondscape(IM), Lake Mogan Pondscape(MO). ***(a)*** TP=total phosphorous *(µg/L)*, ***(b)*** TN=total nitrogen *(µg/L)*, ***(c)*** Chl *a*= Chlorophyll a *(µg/L)*, ***(d)*** DOC=dissolved organic carbon *(mg/L)*.

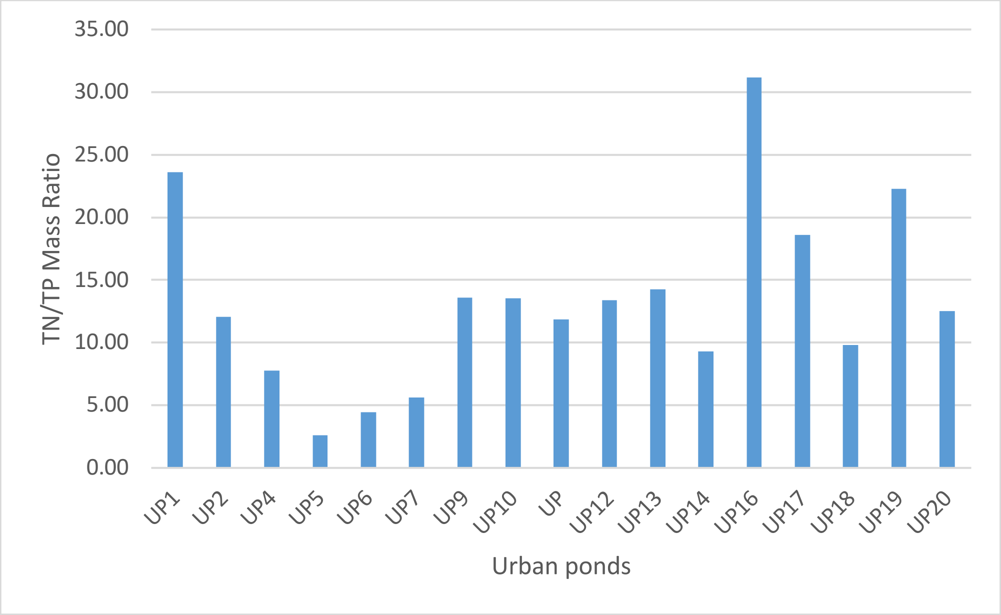

Figure 3.3. The frequency graph showing TN/TP Mass Ratio values in 17 urban ponds (UP). All values ranged between 2.60 to 31.17, and the median is 12.53. Thirteen ponds had lower TN/TP mass ratio values than the remaining.

3.2 Zooplankton Community Composition

Copepods were the most abundant group throughout the urban ponds, while Cladoceras were not that common (see Figure 3.4). The most abundant Copepods were *Cyclops scutifer, Microcyclops rubellus, Nauplii, Copepodid, Macrocyclops albidus,* and *Ergasilus sp*. On the other hand, the most abundant Cladoceran species were *Chydorus sphaericu, Daphnia hyaline, Bosmina longirostris,* and *Moina macrocopa.*

Figure 3.4. The frequency graph showing zooplankton species presence in 17 urban ponds (UP), species were shown on the y-axis, and the number of ponds in which species were present was shown on the x-axis. The red dot represents Cladocera group, while the blue dot represents Copepod group.

The total zooplankton, copepod, and cladocera abundance weighted mean community body sizes for each urban pond were shown in Table 3.3. The lowest body sizes for total zooplankton, copepods, and cladocerans were 0.2793, 0.3747 mm, and 0.2510 mm in order, which was observed in Imrahor River Valley pondscape. On the other hand, the highest body size for total zooplankton and copepods was 1.0979 mm, which was observed in Lake Mogan pondscape, and for cladocerans was 0.5403 mm, which was observed in Gölbaşı Düzlüğü pondscape.

When comparing the pondscapes concerning the average values (see Figure 3.5), the highest average total and copepod abundance weighted mean community body sizes were observed in Lake Mogan pondscape, while the highest average body size for cladocerans was observed in Imrahor River Valley pondscape.

Table 3.3 Zooplankton abundance-weighted mean community body size for 17 urban ponds (UP). PS=Pondscape, IM= Imrahor River Valley pondscape, GD=Gölbaşı Düzlüğü pondscape, MO=Lake Mogan pondscape, total zooplankton mean community body size (mm), copepod mean community body size (mm), and cladocera mean community body size (mm).

PS	***Pond Code***	***Total zooplankton (mm)***	***Copepod (mm)***	***Cladocera (mm)***
IM	UP1	0.4094	0.4291	0.3969
IM	UP2	0.5782	0.5901	0.3760
IM	UP4	0.3719	0.3747	0.2963
IM	UP5	0.4907	0.4938	0.2510
IM	UP6	0.5052	0.5096	0.3098
IM	UP7	0.4595	0.4709	0.2606
IM	UP9	0.8127	1.0568	0.3987
IM	UP10	0.6151	0.6308	0.3170
IM	UP	0.2793	0.5570	0.2708
GD	UP12	0.6888	0.7055	0.5403
GD	UP13	0.4359	0.4757	0.4060
GD	UP14	0.6408	0.6431	0.3780
MO	UP16	1.0979	1.0979	0.0000
MO	UP17	0.8930	0.8930	0.0000
MO	UP18	1.0231	1.0231	0.0000
MO	UP19	0.6421	0.6431	0.4530
MO	UP20	1.0038	1.0038	0.0000

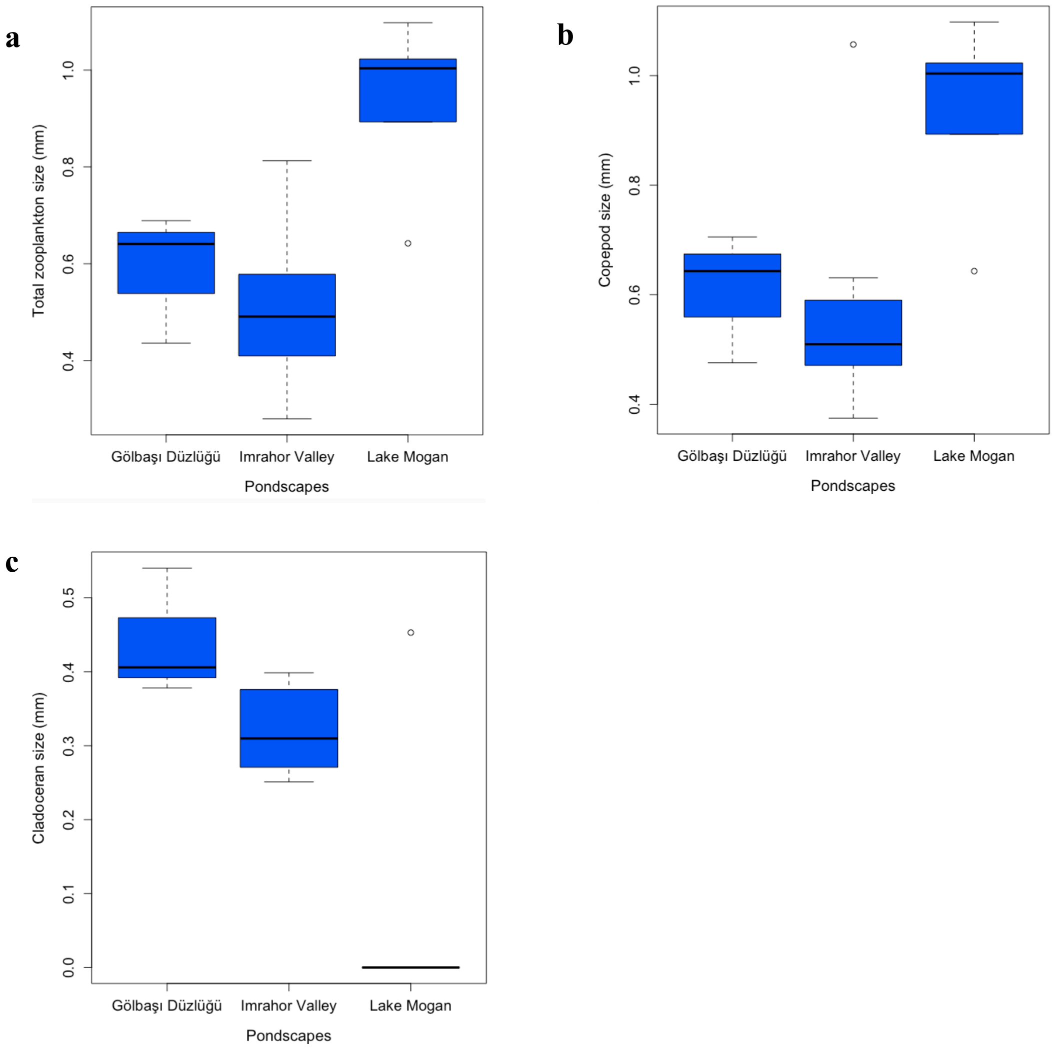

Figure 3.5. Zooplankton abundance-weighted mean community body size of urban ponds (UP) for each pondscapes; Gölbaşı Düzlüğü Pondscape, Imrahor River valley Pondscape, Lake Mogan Pondscape. ***(a)*** Total zooplankton mean community body size (mm), ***(b)*** Copepod mean community body size (mm), and ***(c)*** Cladoceran mean community body size (mm)

The total zooplankton biomass for each urban pond was shown in Table 3.4. The lowest biomass for total zooplankton, copepods, and cladocerans was 2.24 mg, 1.42 mg, and 0.20 mg in order, which was observed in Imrahor River Valley pondscape.

On the other hand, the highest biomass for total zooplankton, copepods, and cladocerans was 42.25 mg, 36.92 mg, and 22.16 mg in order, which was observed in Gölbaşı Düzlüğü pondscape. When comparing the pondscapes concerning the average biomass (see Figure 3.6), the highest average biomass for total zooplankton and cladocerans was observed in Gölbaşı Düzlüğü pondscape, and for copepods was observed in Lake Mogan pondscape. On the other hand, the lowest average biomass for copepods and cldocerans was observed in Imrahor River Valley pondscape, for total zooplankton the other two pondscapes had similar average biomass.

Table 3.4 Total zooplankton biomass for 17 urban ponds (UP). PS=Pondscape, IM= Imrahor River Valley pondscape, GD=Gölbaşı Düzlüğü pondscape, MO=Lake Mogan pondscape, total crustacean zooplankton biomass (µg/L), total copepod biomass (µg/L), and total cladocera biomass(µg/L)

PS	***Pond Code***	***Total zooplankton (µg/L)***	***Copepod (µg/L)***	***Cladocera (µg/L)***
IM	UP1	4.34	1.72	2.62
IM	UP2	5.18	2.50	2.68
IM	UP4	16.76	10.23	6.65
IM	UP5	7.62	7.42	0.20
IM	UP6	8.55	6.99	1.57
IM	UP7	10.17	9.33	0.84
IM	UP9	34.97	29.16	5.81
IM	UP10	12.04	11.29	0.76
IM	UP	2.24	1.42	0.81
GD	UP12	42.25	21.94	20.32
GD	UP13	28.76	6.60	22.16
GD	UP14	12.07	11.52	0.54
MO	UP16	36.92	36.92	0.00
MO	UP17	4.83	4.83	0.00
MO	UP18	14.30	14.30	0.00
MO	UP19	21.90	16.62	5.28
MO	UP20	10.51	10.51	0.00

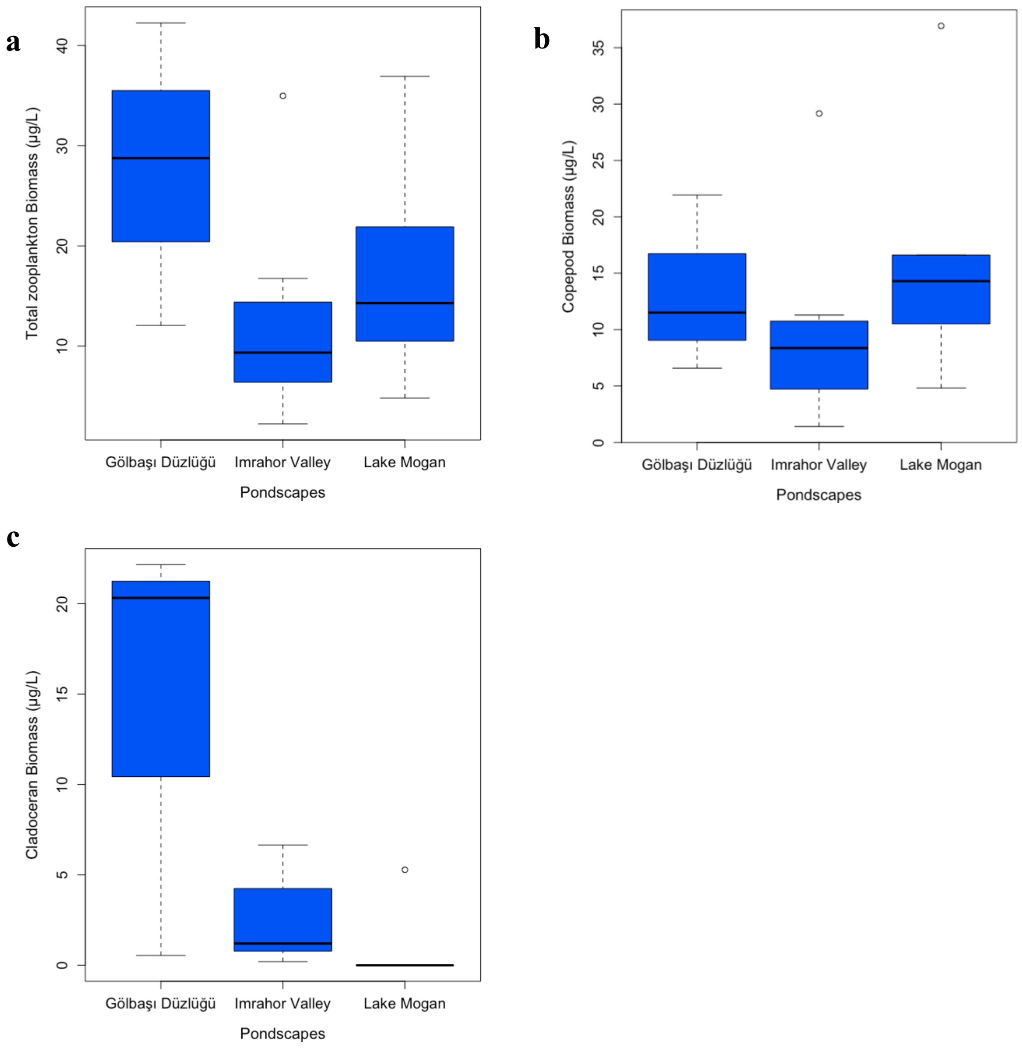

Figure 3.6. Zooplankton biomass of urban ponds for each pondscapes; Gölbaşı Düzlüğü Pondscape, Imrahor River valley Pondscape, Lake Mogan Pondscape. **(a)** Total zooplankton biomass (µg/L), **(b)** Copepod biomass (µg/L), and **(c)** Cladoceran biomass (µg/L)

3.3 Physical and biological drivers of trophic structure (Piecewise SEM)

Results of 6 different piecewise structural equation model for 17 urban ponds in Ankara (Table 3.5). Interactive effects of Chl *a* with TN and TP were not significant for each piecewiseSEM model.

Table 3.5 Descriptive summary statistics of Piecewise Structural Equation model (SEM) of biomass and abundance-weighted mean body size of the zooplankton community. The significance level is set at $p < 0.1$. TP= total phosphorous, TN= total nitrogen, Chl *a*= phytoplankton abundance, Macroinvertebrate= macroinvertebrate abundance, se=standard estimate.

SEM	Significant variables (p<0.1)	Interactions assessed
Total zooplankton abundance weighted mean body size	TN (p=0.0194, se=1.0760), Macroinvertebrate (p=0.0442, se= 0.7370)	TN and Chl *a* (p= 0.5836) interaction was not significant TP and Chl *a* (p= 0.2445) interaction was not significant
Cladocera abundance weighted mean body size	Nothing	Nothing
Copepod abundance weighted mean body size	TN (p=0.0094, se=1.1107), Macroinvertebrate (p=0.0582, se= 0.6243)	TN and Chl *a* (p= 0.8763) interaction was not significant TP and Chl *a* (p= 0.3175) interaction was not significant
Total zooplankton biomass	TN (p=0.0479, se=0.2526), TP (p= 0.0691, se=-0.2396)	TN and Chl *a* (p= 0.6634) interaction was not significant. TP and Chl *a* (p= 0.449) interaction was not significant
Cladocera biomass	Nothing	Nothing
Copepod biomass	TN (0.0275, se=0.1842), TP(p=0.0778, se=-0.0508) Macroinvertebrate (p=0.0626, se=0.0768)	TN and Chl *a* (p=0.7927) interaction was not significant. TP and Chl *a* (p=0.1163) interaction was not significant

3.3.1 Total zooplankton biomass and size

Piecewise structural equation model (PSEM) result was shown in Figure 3.7 and Figure 3.8. According to the model, total zooplankton biomass and abundance-weighted mean community body size were significantly positively related to TN concentration (see Figure 3.9a and Figure 3.9c). Total zooplankton biomass was slightly negatively related to TP concentration (p= 0.0691, se= -0.2396) (Figure 3.9b), but TP concentration was not related to total zooplankton abundance-weighted mean community body size. On the other hand, total zooplankton abundance-weighted mean community body size was significantly positively related to macroinvertebrate predators (*p*= 0.0442, se= 0.7370) (Figure 3.10), but macroinvertebrate was not related to total zooplankton biomass. As shown in the boxplot, there was a distinct difference between low predatory macroinvertebrate abundance and high predatory macroinvertebrate abundance. When predatory macroinvertebrates were higher in abundance, total zooplankton abundance weighted mean community body size was higher. However, there was no relationship between total zooplankton biomass and the total zooplankton abundance weighted mean community body size with Chl *a*, DOC concentrations, and fish, and there was no interactive effect of nutrients with Chl *a* (see Table 3.5).

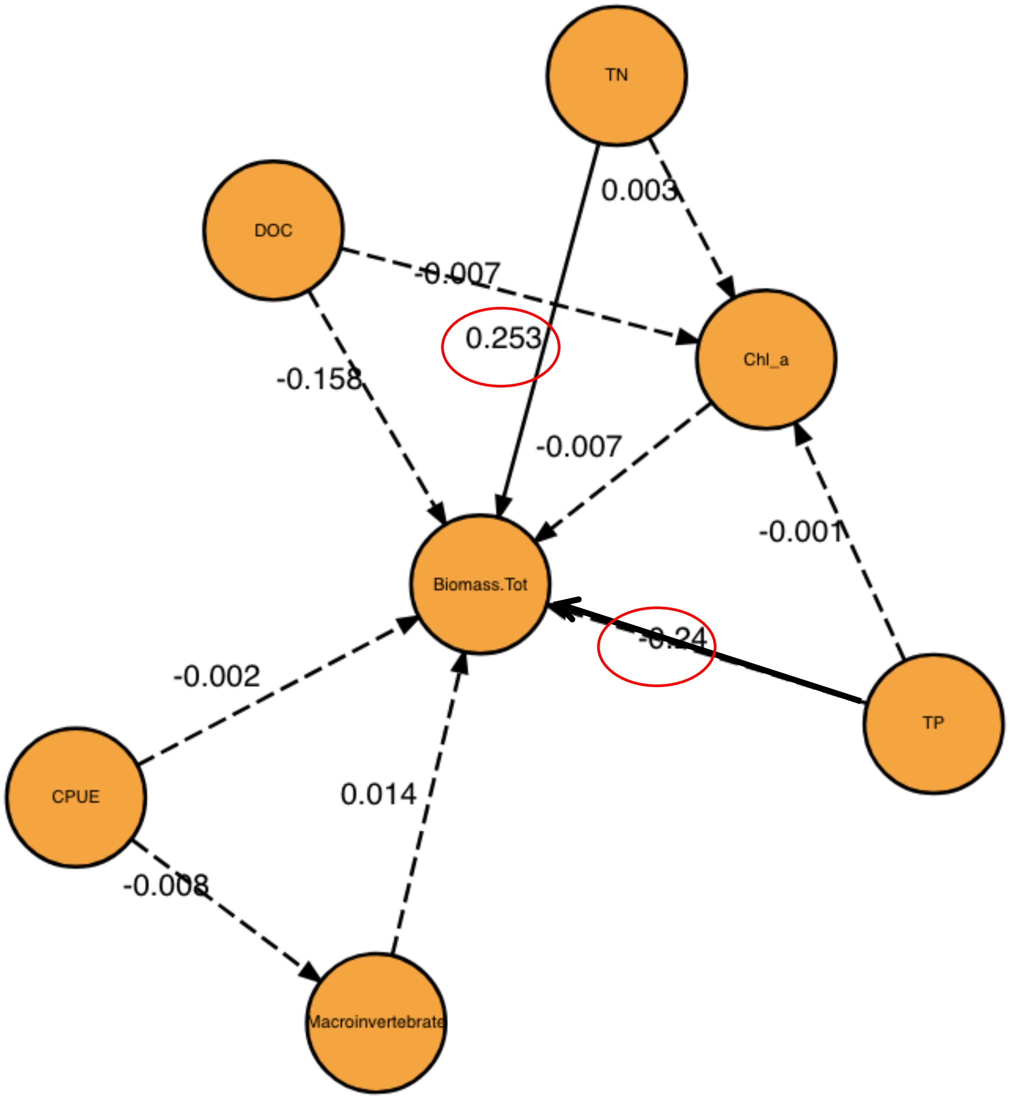

Figure 3.7. Result of PSEM analysis for the total zooplankton biomass. The solid black arrows represent significant paths (p<0.1). Dash arrows represent the non-significant paths, and the standard estimate is shown on the arrow.

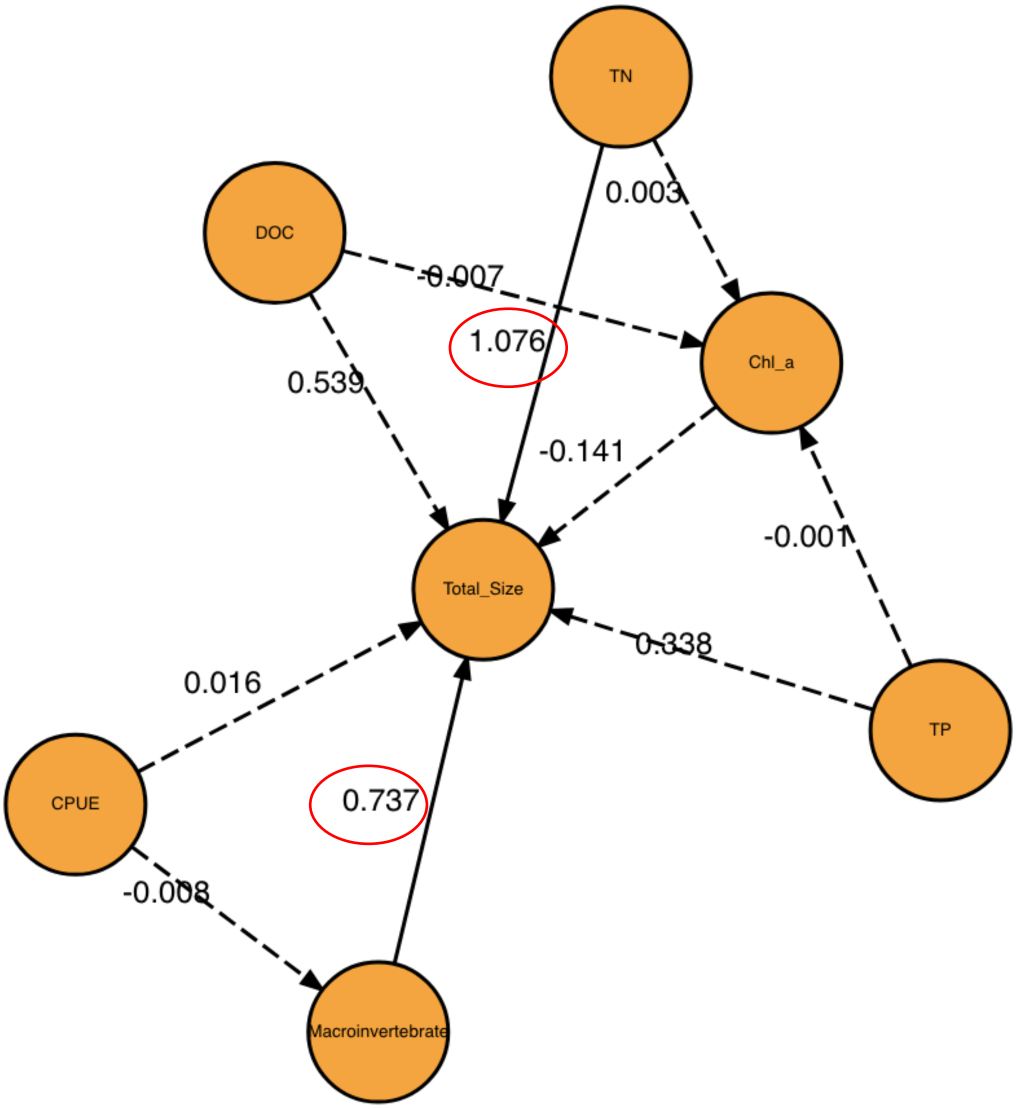

Figure 3.8. Result of PSEM analysis for the total crustacean zooplankton size. The solid black arrows represent significant paths ($p<0.1$). Dash arrows represent the non-significant paths, and the standard estimate is shown on the arrow.

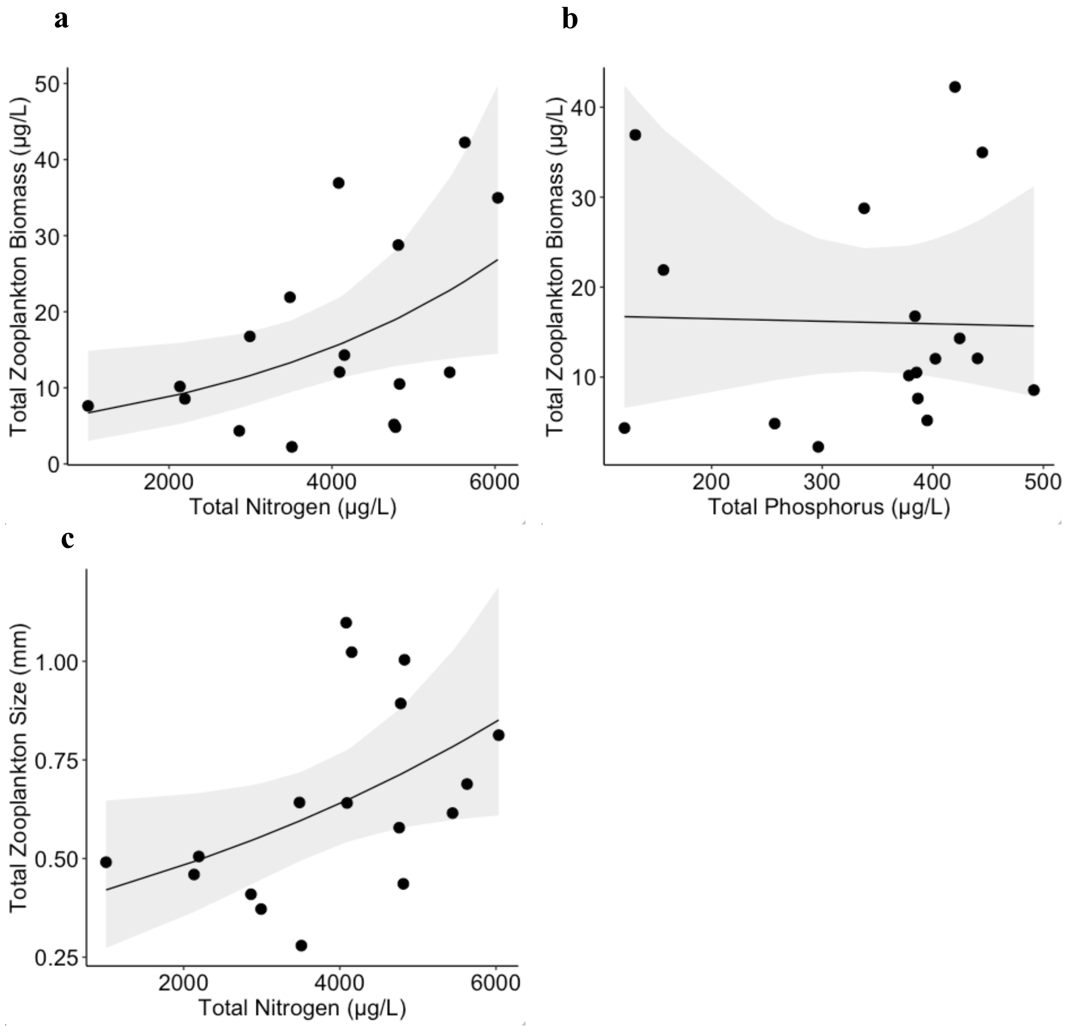

Figure 3.9. Relationship between the total zooplankton biomass and size with nutrients (**a**) the total zooplankton biomass (µg/L) (*y*-axes) and total nitrogen (µg /L), (**b**) the total zooplankton biomass (µg/L) (*y*-axes) and total phosphorus (µg /L) (*x*-axes), and **(c)** total zooplankton size (mm) (*y*-axes) and total nitrogen (µg /L) (*x*-axes) for 17 urban ponds in Ankara. Shaded regions represent the 95% confidence interval estimated from the best-fitting generalized linear or robust regression model.

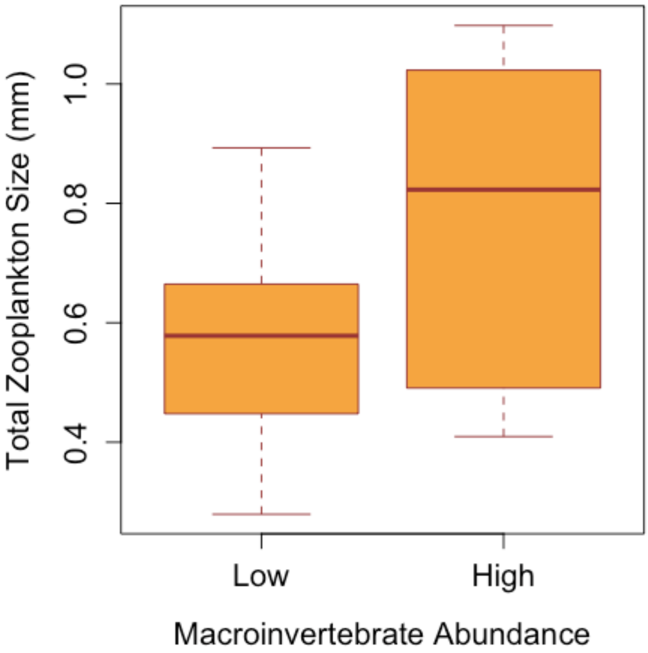

Figure 3.10. Boxplot of the total zooplankton size (mm) (*y*-axes) in 17 urban ponds according to low predatory macroinvertebrate abundance and high predatory macroinvertebrate abundance (*x*-axes).

3.3.2 Copepod biomass and size

Piecewise structural equation model (PSEM) result was shown in Figure 3.11 and Figure 3.12. According to the model, copepod biomass and abundance-weighted mean community body size were significantly positively related to TN (see Figure 3.13a and Figure 3.13c) and Macroinvertebrate (Figure 3.14). On the other hand, copepod biomass was slightly negatively related to TP concentration (p= 0.0778, se=-0.0508), but TP concentration was not related to copepod abundance weighted mean community body size (Figure 3.13b).

Furthermore, copepod biomass and abundance-weighted mean community body size were not correlated with Chl-*a*, DOC, or fish, and there was no interactive effect of nutrients with Chl-*a* (see Table 3.5).

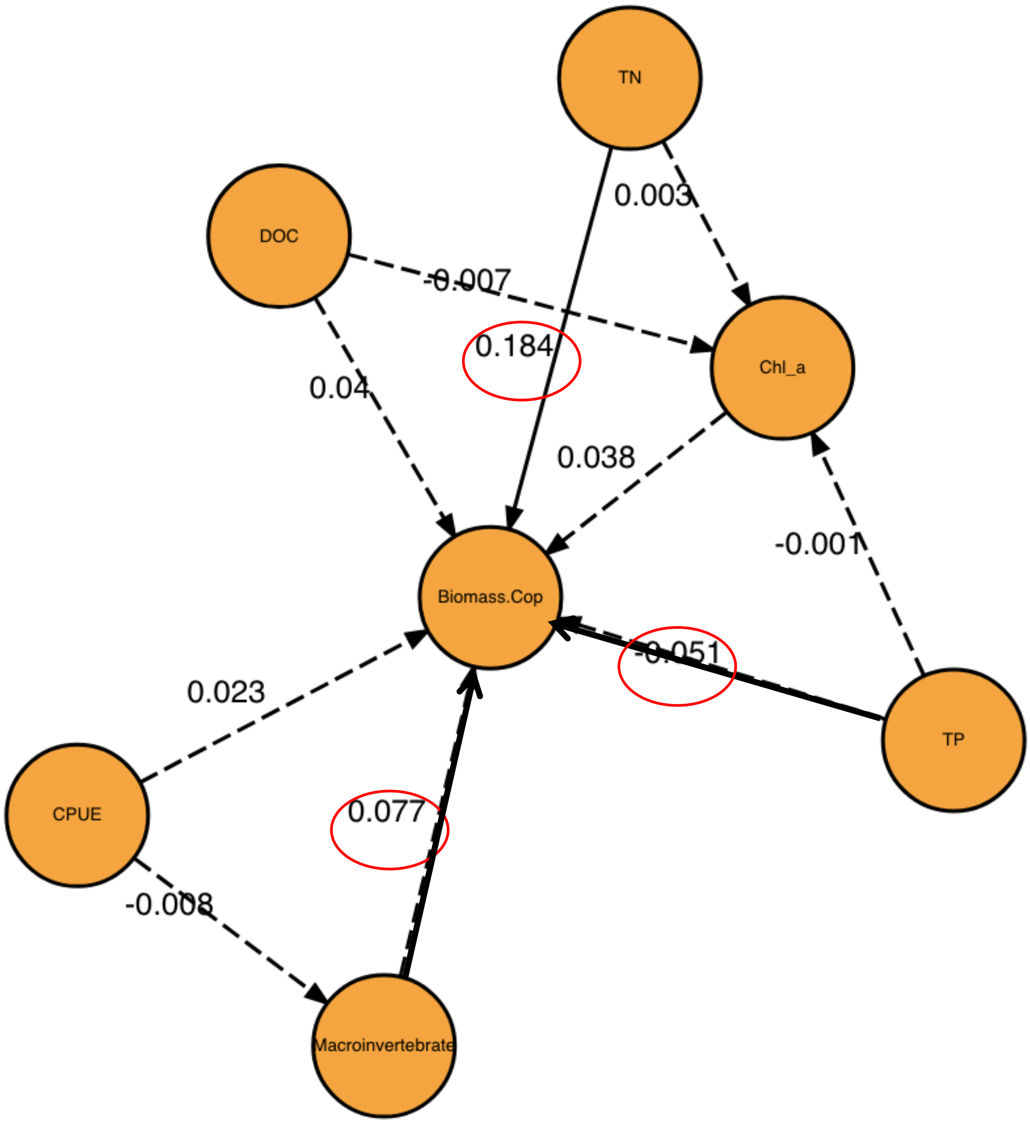

Figure 3.11. Result of PSEM analysis for the copepod biomass. The solid black arrows represent significant paths (p<0.1). Dash arrows represent the non-significant paths, and the standard estimate is shown on the arrow.

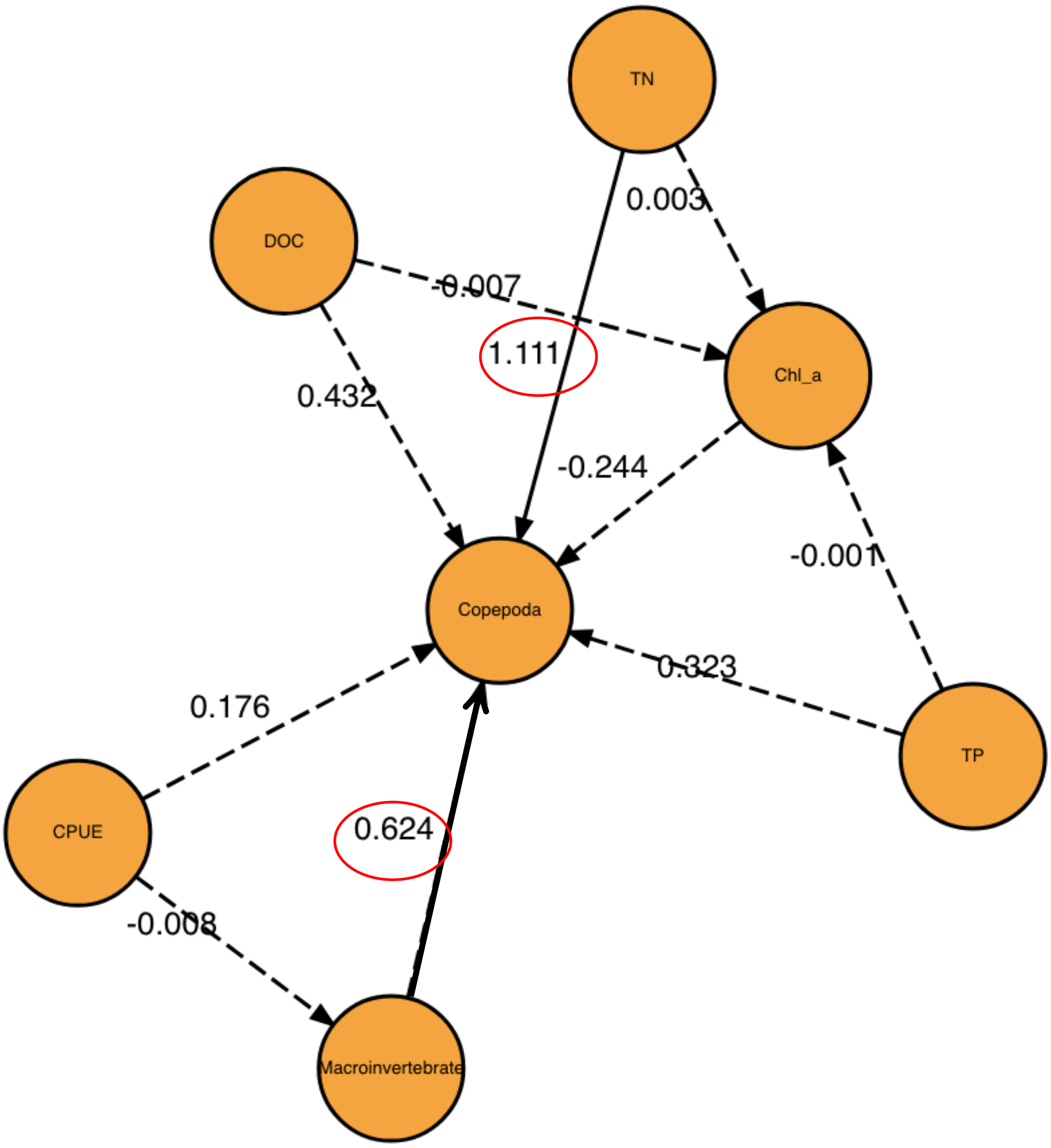

Figure 3.12. Result of PSEM analysis for the copepod size. The solid black arrows represent significant paths (p<0.1). Dash arrows represent the non-significant paths, and the standard estimate is shown on the arrow.

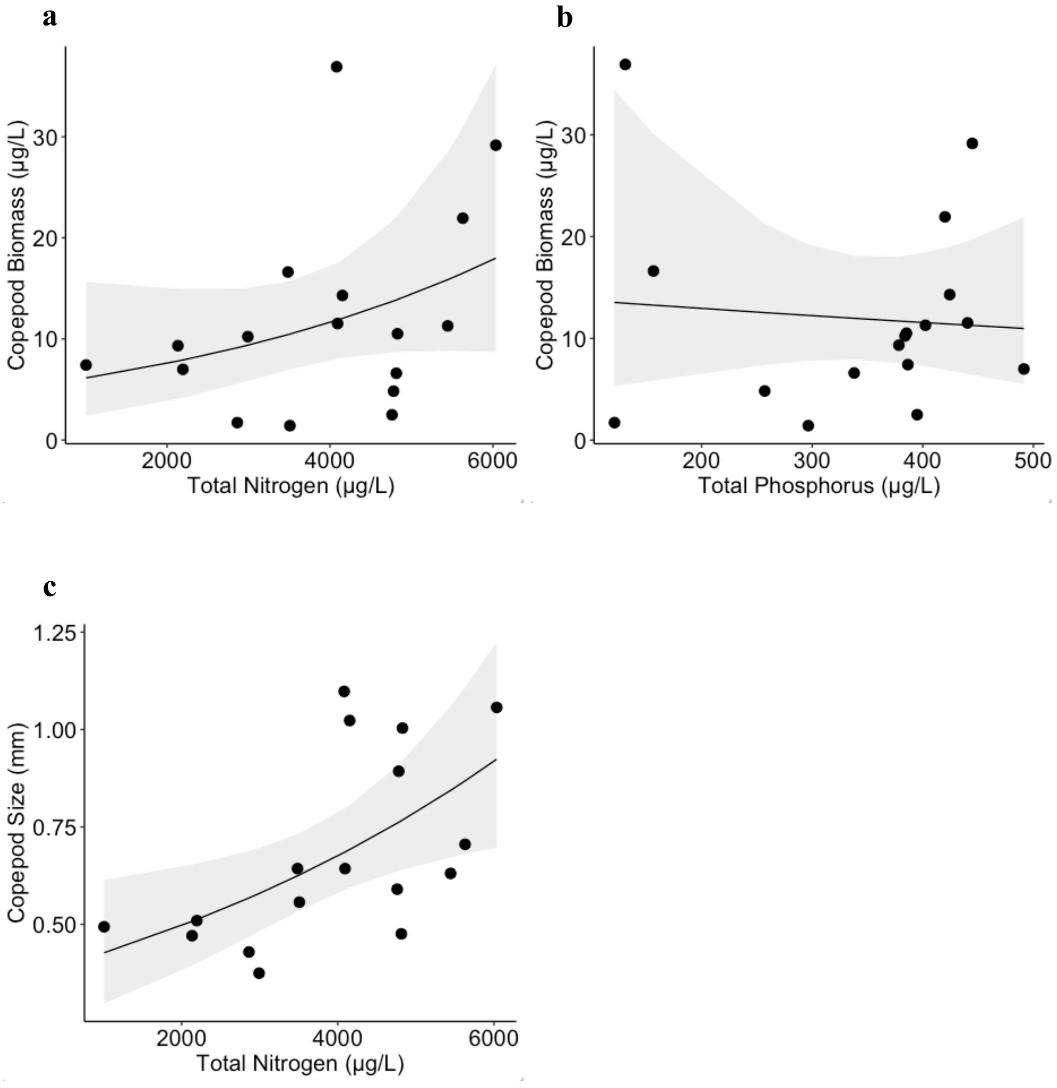

Figure 3.13. Relationship between copepod biomass and size with nutrients (**a**) copepod biomass (µg/L) (*y*-axes) and total nitrogen (µg /L), (**b**) copepod biomass (µg/L) (*y*-axes) and total phosphorus (µg /L) (*x*-axes), and **(c)** copepod size (mm) (*y*-axes) and total nitrogen (µg /L) (*x*-axes) for 17 urban ponds in Ankara. Shaded regions represent the 95% confidence interval estimated from the best-fitting generalized linear or robust regression model.

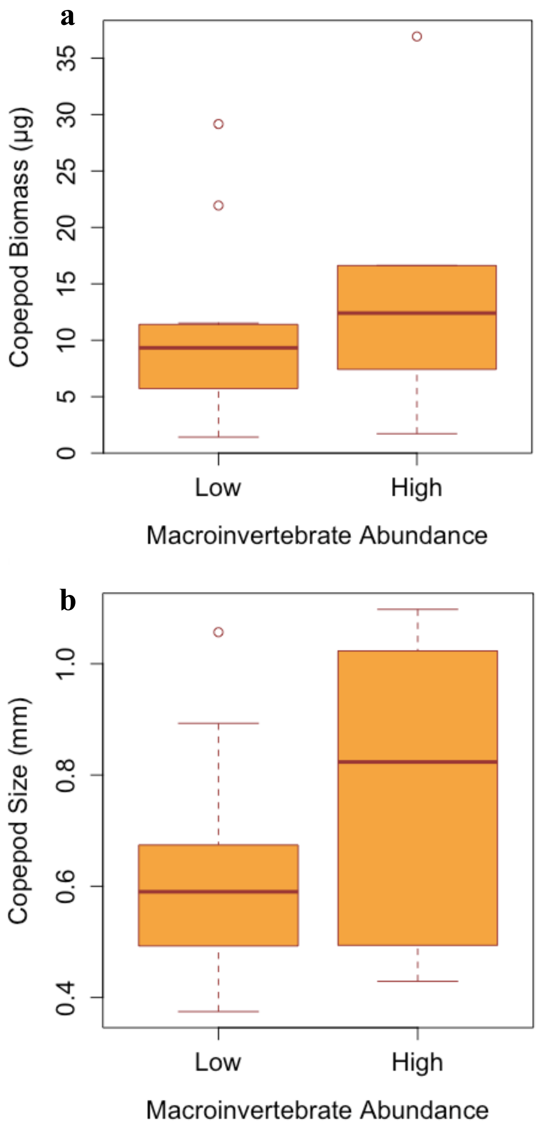

Figure 3.14. Boxplot showing **(a)** copepod biomass (µg) (*y*-axes) **(b)** copepod size (mm) (*y*-axes) in 17 urban ponds according to low predatory macroinvertebrate abundance and high predatory macroinvertebrate abundance (*x*-axes).

3.3.3 Cladoceran biomass and size

Piecewise SEM showed that there were no statistically significant impacts of explanatory variables. Cladoceran biomass was not related to either nutrients or predators. Moreover, Piecewise SEM showed no statistically significant impacts of explanatory variables on cladoceran abundance weighted mean community body size.

CHAPTER 4

DISCUSSION

This study revealed that total zooplankton size, copepod biomass, and copepod size were primarily controlled by bottom-up (nutrients) and top-down (macroinvertebrates), while total zooplankton biomass was controlled by only bottom-up forces. Total crustacean zooplankton was mainly dominated by cyclopoid copepods since cladocerans were not abundant in studied ponds. Piecewise SEM showed no statistically significant impacts of explanatory variables on cladocerans biomass and size. This may be because of the low cladocerans' presence in the ponds. Almost all ponds had copepod species, and the dominant ones were *Cyclops scutifer* and *Microcyclops rubellus*, but 4 ponds did not have any cladoceran. So, copepod dominance in almost all ponds may overrun Cladocera.

Bottom-up controls were represented as nutrients, dissolved organic carbon, and chlorophyll-*a*. Neither chlorophyll-*a* nor dissolved organic carbon showed any statistically significant result in both biomass and size analyses of all zooplankton groups. The possible interaction of nutrients with Chl *a* concentration was checked (see Table 3.5), however; none of the applied statistical analyses revealed an interaction between chlorophyll-a and nutrients. On the other hand, Chl *a* concentration decreased with DOC shown in Figure 5.1 *(in appendix)* as expected, however, there was no statistical significance detected in the structural equation model. This is contrary to the second prediction that water clarity may indirectly affect the biomass and size of the zooplankton community by reducing food availability. So, the second prediction was rejected.

In addition to that piecewise SEM revealed that total nitrogen (TN) positively correlated with zooplankton biomass and abundance-weighted mean community body size, whereas total phosphorous (TP) concentrations were slightly negatively related to the biomass of total zooplankton and the biomass of copepods but not to the abundance-weighted mean community body size of any zooplankton groups. This is probably because TN:TP mass ratio values showed that most of the urban ponds might be N-limited (see Table 3.2; Figure 3.3) since the mass ratio is lower, while four of them (UP1, UP16, UP17, and UP19) might be co-limited or P limited. TN/TP mass ratio for these ponds is over 18. Also, even though TP seems higher for these ponds; 121.2 µg /L, 130.9 µg /L, 257 µg /L,156.4 µg /L in order, SRP values for these ponds were slightly lower, 64.5 µg /L, 96.4 µg /L, 111 µg /L, 20.3 µg /L in order. So, phosphorus that might be directly taken up by algae was not abundant in these ponds. On the other hand, for the remaining ponds, they were possible that N-limited and had a high TP value. Even though phosphorous showed a small negative indirect effect on zooplankton biomass, it is negligible, as shown in Figure 3.9b and Figure 3.13b, almost a linear, not a detectable direction.

This is just an indication of limiting nutrients because the TN:TP mass ratio threshold for nutrient limitation greatly differs in the literature. For example, Sakamato (1966) stated that when TN:TP mass ratio was >17, phytoplankton biomass was dependent on TP, when TN:TP mass ratio was <10, phytoplankton biomass was dependent on TN, and when TN: TP mass ratio was between 10-17, phytoplankton biomass depended on both.

In this study, when DIN and TN were compared (see Table 3.2; Figure 5.3 in appendix), even though TN concentration was higher, DIN was not that high, showing most of the nitrogen coming from organic sources and might not be easily available to the organisms. Since all DIN forms can be assimilated by phytoplankton, micro and macroalgae, and bacteria, primary producers might have faced N limitation because of low DIN concentrations. Many studies demonstrated a strong positive effect of TN on zooplankton biomass in pelagic food webs (Yan, 1986; Jeppesen et al., 2000).

Shurin et al. (2012) also showed that in experimental freshwater ponds in western Canada, zooplankton biomass was positively correlated with nutrient loadings because of the increased biomass of primary producers. As most ponds were TN limited based on the ratio of low TN:TP and low DIN concentrations, ponds with high TN concentrations were likely to lead to higher primary production that positively and indirectly affected the zooplankton biomass and size even though there was no interaction effect between TN and Chl *a* concentration. So, the positive relationship of TN with biomass and the abundance-weighted mean community body size of zooplankton might have been because of possible N limitation, nitrogen seemed to be the key nutrient in this study, supporting the first prediction that nutrient availability may increase the productivity of the ponds, indirectly affecting the biomass and size of zooplankton positively.

Since top-down and bottom-up processes do not only include pelagic organisms, but also benthic ones, they also contribute to the primary (periphyton, macrophytes, phytoplankton), and secondary productivity of the ponds (Vadeboncoeur et al., 2002). Studies showed that zooplankton can use carbon sources other than phytoplankton, including dissolved organic matter, allochthonous materials, and bacteria (Salonen& Hammar, 1986; Wylie & Currie, 1991; Hessen, 1992; Tranvik, 1992; Grey, Jones & Sleep, 2001). Periphyton also serves as a food source for filter feeders (invertebrates, and zooplankton). In the current study, most of the copepods were cyclopoid, which is raptorial feeders, but some also feed on algae i.e residing both on pelagic and benthic so TN might have affected the biomass and size of the zooplankton community not through phytoplankton, but indirectly through periphyton, and benthic algae. Benthic organisms might have benefited from sunlight, i.e, benthic productivity might have been promoted by higher Secchi depth, reaching the bottom of the pond, and increasing photosynthesis. In the current study, Secchi depths were lower for the deeper ponds (see Table 3.1), while higher for shallower ones. Especially in UP1, UP16, UP17, UP18, and UP20 Secchi depths were the highest *(see* Figure 5.2 *in appendix*), almost the same as the maximum depths, these ponds might have had high productivity in the benthic zone.

In 7 of 17 urban ponds in the current study, planktivorous fish were not found. On the other hand, predatory macroinvertebrates were present in all ponds; in 6 ponds, invertebrates were more abundant than the rest. The result of this study showed that macroinvertebrates were positively correlated with abundance-weighted mean community body size but not correlated with the biomass of the total zooplankton.

For copepods, macroinvertebrates showed a positive effect not only on their size but also on copepod biomass, too. Since fish did not show a statistically significant relation with biomass and size of the zooplankton community, the third prediction cannot be supported.

Most of the studies showed that predatory macroinvertebrates had been shown to directly reduce the zooplankton biomass, including *Notonecta* (Arnér et al., 1998; Shurin,2001), *Chaoborus* (Vanni & Findlay, 1990), and *odonates* (Burks et al.,2001). However, this study revealed the opposite, it was associated positively with the biomass of copepods. There were 3 temporary fishless ponds among the pods studied; they dried out in the summer, and 7 ponds were without fish in total. Statistical analysis did not show any significant role of fish in all urban ponds; it might be because their limited presence was insufficient to show any correlation between either size or biomass of the zooplankton. For that reason, macroinvertebrates might have gained a significant role in shaping the community structure. Cobbaert (2010) demonstrated such a similar case in fishless pond ecosystems in north-central Alberta, Canada *Dytiscus alaskanus* (Coleoptera: Dytiscidae) predation on other predatory macroinvertebrates such as *corixids*, *Chaoborus*, and *Zygoptera* indirectly induced top-down effect by eating of predatory macroinvertebrates which led to an increase in zooplankton biomass. Many studies also show that in a temporary fishless pond, macroinvertebrates became top predators, like notonectids (backswimmers), dytiscids(diving beetles), and dragonflies. In this current study, Dytiscidae was abundant in UP1,UP2, UP, UP12, UP16, UP19, and UP20, and within these ponds except UP, UP16, and UP19, none of them had fish.

Therefore; Dytiscidae might have indirectly induced top-down effect on other predatory macroinvertebrates, and might have relieved predation pressure on zooplankton. So, this might have caused an increase in copepods' biomass.

On the other hand, in fishless lakes and ponds, such as UP1, UP2, UP10, DP12, UP14, UP17, and UP20, the dominance of larger zooplankton may be explained by being superior competitors for resources and being able to grow and reproduce even at lower food concentrations, as in the size efficiency hypothesis, or predation by macroinvertebrates such as *Chaoborus* may cause the removal of smaller zooplankton, and shifting to larger individuals (Dodson, 1972; Hall, 1976; Zaret, 1980). This may explain the fourth prediction that when mouth-gape-limited macroinvertebrates are abundant in the ponds, it may cause predation pressure on smaller community body-sized zooplankton and shift the community to larger body-sized zooplankton and increase the biomass of copepods. *Chaoborus* was not encountered in the current study, this might be because of the diel vertical migration pattern of the species; they might migrate to the sediment during the day to avoid fish predation (Dawidowicz, 1993) in ponds UP4, UP5, UP6, UP7, UP9, UP, UP13, UP16, UP18, and UP19, as sediment samples were not taken, organisms would not have been detected.

Furthermore, dissolved organic carbon (DOC) levels in studied ponds differed greatly and were extremely higher; the average DOC was 29.3 mg/l, with a range between 4.0 mg/l to 79.8. Even though dissolved organic carbon (DOC) did not show any statistical significance, it may indirectly impact the zooplankton community structure. Generally accepted dissolved organic carbon (DOC) for urban ponds is not certain, however, there are some studies for lakes and retention ponds such as Wetzel (2010) studied 500 lakes and retention ponds and showed that dissolved organic carbon (DOC) concentration for oligotrophic and eutrophic lakes ranged between 2.0 mg/L to 10.3 mg/L. Also, Sobek et al. (2007) showed from 7514 lakes on six continents, the average dissolved organic carbon (DOC) was 5.7 mg/L.

Wissel (2003) demonstrated that highly colored water which is because of high dissolved organic carbon (DOC), might cause poor prey perception of planktivorous fish, followed by reduced predation pressure on *Chaoborus* and zooplankton. Therefore, *Chaoborus* became more abundant, and since they are mouth-gape limited and, feed on smaller zooplankton, causing a shift from small species, such as *Bosmina* and small copepods, to larger species such as *Daphnia* and *Holopedium*.

However, in the current study, bigger body-sized zooplankton was represented mainly by copepods since cladocerans were not abundant and big even though many of the ponds were rich in high dissolved organic carbon (DOC), and under the brownification effect. Copepods might have had the chance to get bigger and become more abundant since predation pressure was more on the smaller ones.

4.1 Limitations of this study

This study was conducted to explore the main drivers of the zooplankton biomass and abundance-weighted mean community body size in 17 urban ponds in Ankara, which were never studied before. Sampling except for fish was performed during the summer of 2021. In the early autumn of 2021, fish sampling was performed. Sampling with high temporal resolution would have been more explanatory to show the trophic structure of these urban ponds.

There were a couple of obstacles that were faced during this study. The first was the absence of rotifers' biomass and size measurements, which would have been good to define the zooplankton community on a broader perspective. Without rotifers, counting as total zooplankton was not very appropriate since community representation was limited.

Secondly, zooplankton samples degraded faster due to high debris content. The high density of debris may be caused by the mixing of the sediments with cattle entry or by high land use near the ponds, such as unloading the waste from the brick factory into the ponds, which might have caused this pollution.

Even though samples were filled with 4% Lugol iodine solution, a bad odor, and degradation were detected. May be Lugol solution was not prepared well. To save the samples, 70 % glutaraldehyde was added to each sample, then smell and degradation stopped, but the damage was already done to some individuals. This caused a decrease in the quality and quantity of the zooplankton identification.

Thirdly, the collection of macroinvertebrate samples; the inefficient use of the sampling net caused low aquatic macroinvertebrate abundance, also this method was mainly qualitative, a more quantitative sampling protocol should have been used. Also, the preservation of samples with a lot of debris caused fast degradation of the organisms, and lower identification quality. Moreover, sediment samples were not taken, and sampling was only done during daylight, *Chaoborus* was not detected in identification. This might cause an underestimation of the macroinvertebrate predators' abundance in studied ponds. In addition, sediment samples would be helpful to understand the benthic zone and its role in the trophic cascade.

Lastly, if phytoplankton species of these ponds had been identified, that would have helped to show food availability for zooplankton from a better perspective, and show types of cyanobacteria and their abundance, which would define organism' role better in the trophic cascade.

CHAPTER 5

CONCLUSION

This study assessed the main drivers of the zooplankton biomass and abundance-weighted mean community body size. The explanatory variables were chosen from the GLM and histogram plots. The explanatory variables were TP, TN, Chl *a*, DOC, macroinvertebrates, and fish; the response variables were biomass and abundance-weighted mean community body size. Then, Piecewise SEM (structural equation modeling) was used to show the relationship between the explanatory and response variables. For each response group, piecewise SEM was constructed, showing direct and indirect relationships in the system.

For the total crustacean zooplankton and copepods, TN showed a positive relation with biomass and size. This might indicate the nitrogen limitation of the ponds, and the possible importance of the benthic zone in the trophic structure. As TN/TP mass ratio values and low DIN values showed, most urban ponds might be N-limited. While TP showed a small negative relationship only with biomass, and it is negligible, not a very strong relationship. Additionally, macroinvertebrates showed a positive relationship with total zooplankton and copepod size and a positive relationship with copepod biomass. For cladoceran, none of the explanatory variables showed a statistically significant result, this might be because of the low number of cladoceran density across all urban ponds furthermore, in 4 ponds cladocerans were not encountered.

These findings support the two hypotheses that nutrient availability and predation by macroinvertebrates significantly affect zooplankton community structure in urban ponds in Ankara.

APPENDIX

A. Additional Figures

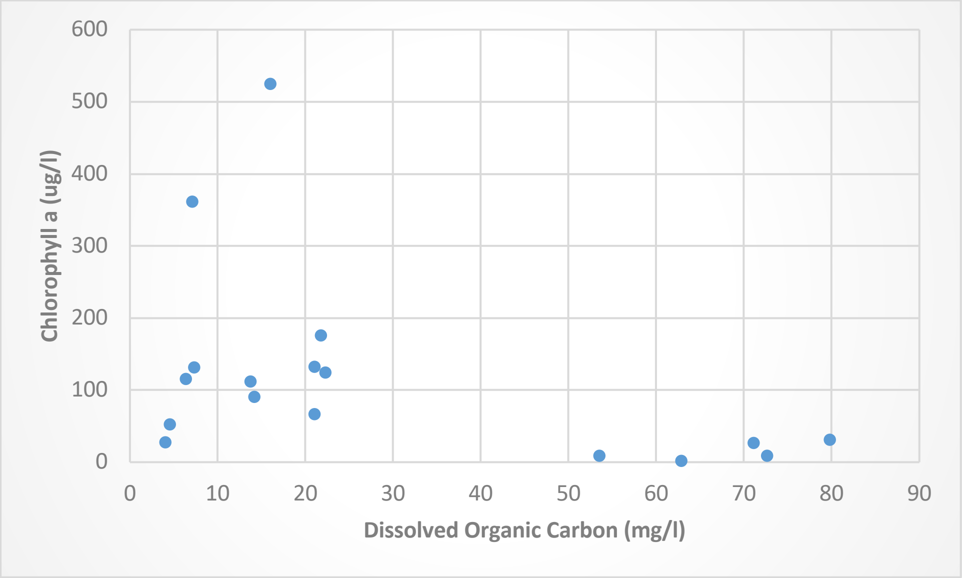

Figure 5.1. Relationship between Chlorophyll *a* (µg/L) (*y*-axes) and Dissolved organic carbon (mg/L) (*x*-axes) for 17 urban ponds in Ankara.

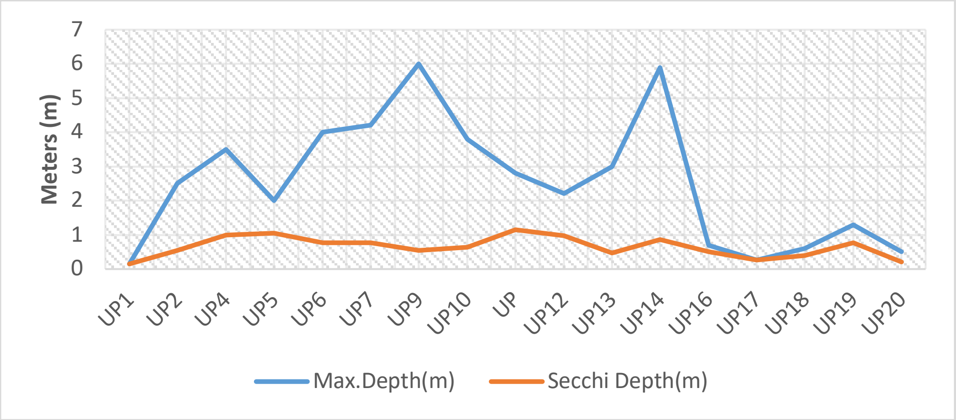

Figure 5.2. Depth and Secchi depth values of 17 urban ponds (UP). The blue line represents the maximum depth (m), and the orange line represents the Secchi depth (m). Higher Secchi depth values were observed in Lake Mogan pondscape, while fluctuations were observed in the other two pondscape.

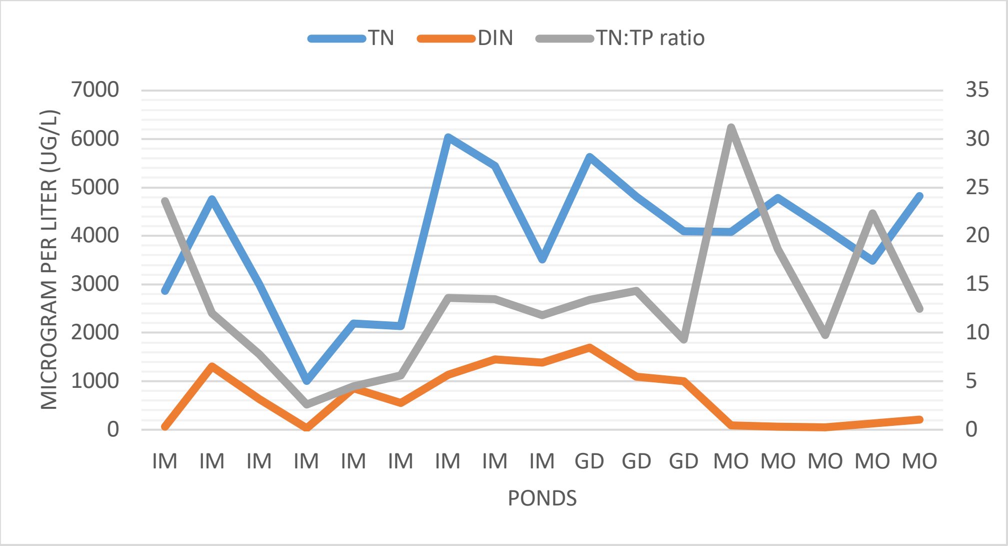

Figure 5.3. Total Nitrogen (TN), Dissolved Inorganic Nitrogen (DIN), and TN:TP Mass Ratio for 17 urban ponds for each pondscapes; Imrahor River valley Pondscape (IM), Gölbaşı Düzlüğü Pondscape (GD) , Lake Mogan Pondscape (MO).

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