Application of Diatom Analysis to Water Quality Assessment. Case Study: The Nakdong River in the Republic of Korea

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Abstract

It is important to maintain a good level of water quality and river health in ecosystems because water is of the essence for humans and any living organism. Thanks to the advantages of diatoms, diatom-based indices are used to assess river health but a short-term assessment using one year of monitoring duration and one to two samples per year in the frequency is dominant. As a result, a holistic understanding of river health on a long-term scale lacks and the changes in diatom assemblages have been poorly understood despite the amass of the government diatom datasets in South Korea.

The Nakdong river, one of the largest rivers in South Korea, has eight barrages, constructed during 2010–2012, which control the river flow using sluices. This project aims to understand the relations between water quality and diatom assemblages regarding the construction of the barrages in the Nakdong river and its tributaries over the period 2009–2018.

This research demonstrates a holistic understanding of river health by looking at changes in water quality and diatom assemblages over ten years using species composition and multivariate analyses. Findings show that the barrages have caused significant changes to the flow regime and thermal system in the main river through impoundment and increased water levels with worsening trends of water quality. The construction has also triggered the significant changes in diatom assemblages in the main river, while the tributaries remain similar. In these findings, diatoms are central to the link between hydrological change, water quality, and the construction of the barrages. This study is a prime example of river health assessment on a multiple-years scale by providing a whole picture of river health in time and space.

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Declaration

I confirm that no part of the material presented in this thesis has previously been submitted by me or any other persons for a degree in this or any other university. In all cases, where is relevant, material from the work of others has been acknowledged.

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Statement of Copyright

Acknowledgements

It is still feeling weird to find myself writing acknowledgement in my PhD thesis. Looking back last four years and four months of my PhD journey, it has never been easy, dealing with lots of small problems and lots of big challenges along the way on top of the global pandemic. However, I have made it in the end and I am so proud of myself.

I still vividly remember the moment when I was travelling to the U.K. in September 2017. I embarked on this journey with a desire to grow into a scientist. I was excited to make this whole new journey, but at the same time, I was incredibly feeling nervous about new challenges and uncertainties in this journey and how I could cope with them.

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Sincerely
Muyeol Jung
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CHAPTER 1

Introduction

1.1 Rationale

Catchment systems store and transfer precipitation inputs and the rivers within them act as pathways that connect the land with lakes, or seas, and flow between associated ecosystems. Due to intensive anthropogenic activities such as agriculture, industrialisation, and urbanisation, major rivers in the world have been severely impacted by nutrient enrichment and modification (Anderson et al., 2002; Oliveira and Machado, 2013; Glibert, 2017). The over-enrichment of water by nutrients such as nitrogen and phosphorus in runoff or effluents results in eutrophication and the deterioration of water quality in rivers, which subsequently can cause hypoxia as well as harmful algal blooms, posing significant threats to humans as well as aquatic biota living in rivers (Boyd, 2019).

Hence, it is one of the top priorities for environmental engineers to adequately tackle water pollution in rivers, maintain water quality, and restore river environments to healthy states. In general, a lot of efforts have been made to accomplish these goals, for example, with the establishment of water quality monitoring programmes, the instalment of sewage plant facilities, and the introduction of legislation to promote appropriate river management. Furthermore, with emphasis on an ecological perspective or even a whole-systems perspective (e.g. the EU Water-Framework Directive), there is growing interest in river restoration to restore a river to a former or original condition by removing artificial barriers in a channel or reconstructing meandering channels and wetlands (Palmer et al., 2005).

Nevertheless, water pollution remains one of the major environmental problems that humans face and the ecosystem in large rivers all over the world is at risk with the loss of biodiversity (Glibert, 2017). Moreover, climate change makes it more challenging to deal with environmental issues moving forward. Increased temperature and changes in precipitation patterns over the globe will considerably affect the hydrologic and thermal regimes of rivers, subsequently making a direct impact on water quality and freshwater ecosystems (Vliet et al., 2013). The IPCC

(2014) warn that a large portion of territorial, freshwater, and marine species would face an increased extinction risk due to climate change.

Under these circumstances, the term 'river health' gained popularity in the late 1990s. River health is often seen as being analogous to human health and can be defined in various ways, depending on what aspects of river health are measured and assessed, from physical characteristics to chemical characteristics, to even the societal values of rivers (Karr, 1999; Norris and Thoms, 1999). Despite inherent problems with the metaphor that health is not an observable ecological property, and the term being unclear on what aspects of river health are assessed (Suter, 1993; Norris and Thoms, 1999; Blue, 2018), it became widely accepted and used in riverine ecological studies, owing to advances in the biological assessment and development of biological indices.

In recent years, most water quality research combines the physical, chemical, and biological aspects of rivers for a more integrated river health assessment. Traditionally river health was primarily assessed using water quality indices based on the physical and chemical measurements of water in rivers, focussing on things such as temperature, suspended solids, pH, dissolved oxygen, and nutrients. However, the measurement of water may change diurnally, nocturnally, daily, seasonally, or annually and it is difficult to understand the actual meanings of such measurements in relation to the ecosystem. Besides, it is costly and onerous to set up water quality monitoring stations across a catchment and constantly keep on recording them over time.

Thus, biological indices using epilithic diatoms, invertebrates, and fish became popular as tools to assess river health. One of their strengths is that they can reduce the complex array of ecosystem responses to simple intuitive scores (Suter, 1993). However, it can be difficult to understand the meaning of these results, because the responses of aquatic biota can be dependent on more than one component. Thus, it is difficult to determine the reasons for high or low values.

River health assessment lies at the intersection of various disciplines from physical geography to biology, ecology, and hydrology and various elements simultaneously interact with each other (Poole, 2002). Therefore, for an accurate assessment of river health using aquatic biota, an extensive examination of relationships between environmental variables that affect aquatic biota, such as habitat structure, flow regime, energy sources, water quality, and biological interactions and biological conditions is strongly required in the study of river health (Norris and Thoms, 1999).

Despite these challenges, biological monitoring is essential in river health assessment as it provides useful tools that enable the identification of biological responses to human actions. Ultimately, the understanding of biological responses in rivers will not only help raise public awareness of environmental problems but also guide environmental managers in the right direction in river management planning. Furthermore, a wide range of biological responses to changes can be used as criteria for river rehabilitation projects.

1.2 The Nakdong river in the Republic of Korea

1.2.1 Construction of barrages in the Nakdong river

Behind the successful economic growth of South Korea, water pollution in rivers and recurring algal blooms are some of the biggest environmental challenges the country is facing. Influenced by monsoonal precipitation and typhoons, 70 % of South Korea's annual precipitation is concentrated over summer and early autumn (Korea Meteorological Administration, 2012). Such an imbalance in precipitation means that catchments are prone to flooding and droughts, making it extremely difficult for the government to securely maintain water resources throughout the year. In addition, with increasing trends of temperature and precipitation expected in South Korea due to the effects of climate change (Boo et al., 2006), there was a strong need for the government to take proactive action to prepare for extreme climatic events.

Accordingly, in July 2009, as part of the Four Major Rivers Restoration Project (FMRRP) (Kim, 2009) whose aims were 1) to prevent flooding, 2) to secure abundant water resources, 3) to improve water quality as well as river health, and 4) to provide recreational spaces on the riverside, eight barrages* have been constructed along the main channel of the Nakdong river which control the river flow using sluices (Figure 1.1). The project involving the construction of the barrages, dredging of the river channel, and removal of sediment bars in the channel was completed in April 2012 (The FMRRP Committee, 2014).

Despite the completion of the project, harmful algal blooms continued to occur in the Nakdong river with an increase in frequency and severity, which raised a question on the effects of the project (Park, 2012). In fact, the project was immediately met with a huge backlash from environmentalists and conflict increasingly intensified, leaving the country profoundly divided over whether the project has actually improved water quality and river health as intended or not (See Normile, 2010; Shin and Chung, 2011; Lah et al., 2015 for the FMRRP). Furthermore, there is ongoing debate over whether the barrages should be demolished to restore the previous state of the river because of increased occurrences of algal blooms, concerns over the ecological impacts and the large costs for maintenance of the barrages and its facilities.

In spite of the large scale of the construction across the main river, there was little consideration at the planning stage on the environmental and ecological impacts. In the post-construction period, some research has been carried out on hydrologic, biological, ecological, and geomorphological changes in comparison to

*Although, the barrages constructed in South Korea have watergates, most Korean literature adopts the term "weir" instead of "barrage" in their studies. In this thesis, however, barrages are used for the artificial barriers that were constructed as a result of the FMRRP and the two terms are separately used because of their different roles on river flow according to the dictionary of civil engineering (Scott, 1991). It defines "barrage" as "a low dam gated across its entire width, placed across a river to raise its level" (p. 27), while "weir" as "a wall built across the full width of a stream with a horizontal crest over which the water flows" (p. 494).

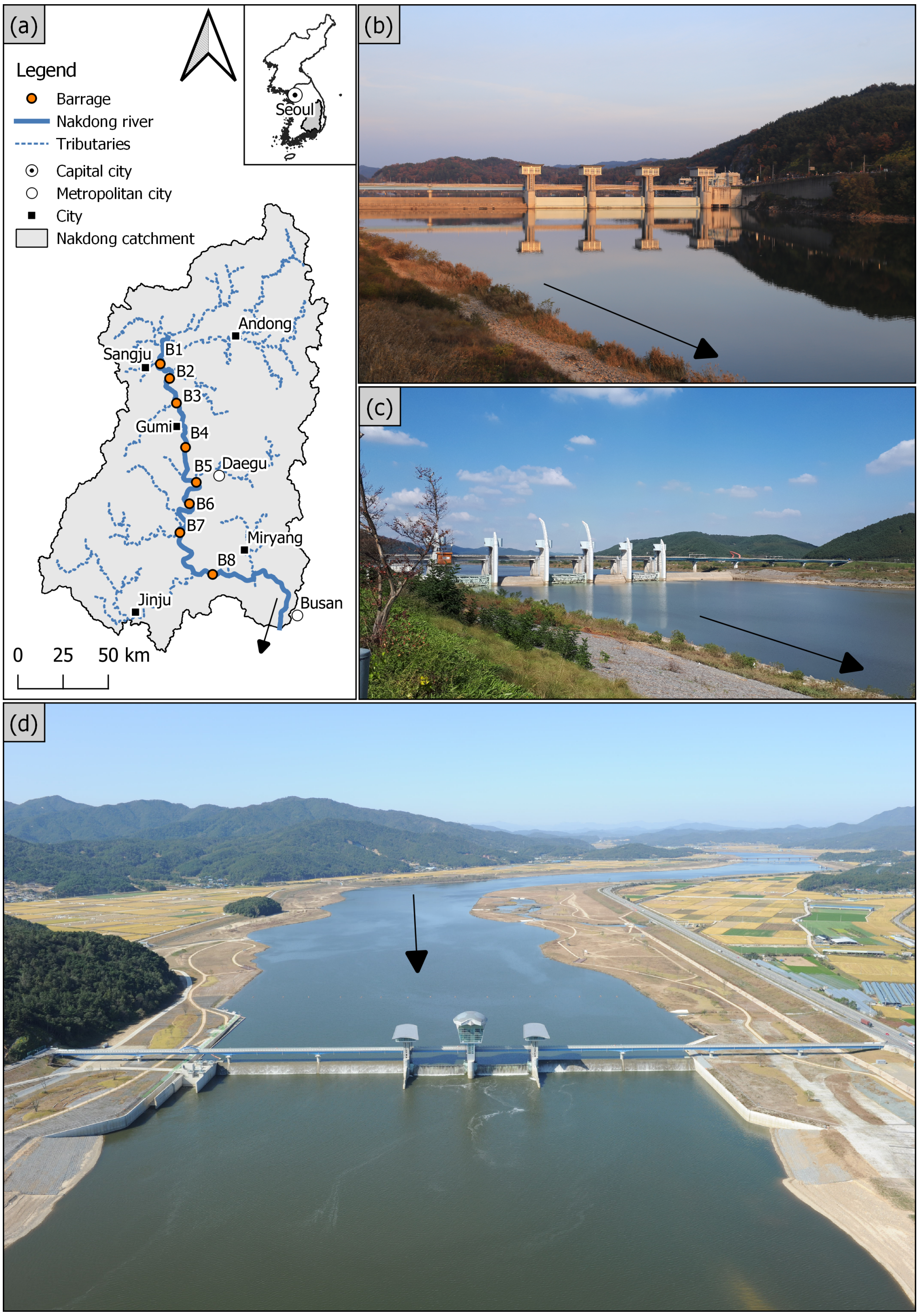


Figure 1.1: Location of eight barrages in the Nakdong river. (a) The location of eight barrages constructed along the Nakdong river in the study area, (b) A photo of Nakdanbo Barrage (B2), taken on 6th Nov 2018, (c) A photo of Hapcheon-Changnyeongbo Barrage (B7), taken on 18th Oct 2018, (d) A photo of Gumibo Barrage (B3) taken by K-Water, 2015. Arrows on the images indicate the flow direction.

the pre-construction period state, but there is a shortcoming in the literature that none of research has taken a comprehensive and long-term approach.

1.2.2 Geographical setting in the Nakdong catchment

The study area, the Nakdong catchment, is an area of 23,690 km^2 and lies in the southeast region of South Korea where approximately 6.6 million people reside, mainly relying on the Nakdong river and its tributaries for water resources (Water Resources Management Information System, 2019) (Figure 1.2). The basin is separated by the two large mountain ridges (altitudes $>$ 1000 m) running from the north to the south on the east and west sides of the basin, whereas the southernmost boundary is formed by low altitude mountains and hills (Figure 1.2 (b)). Within the catchment, most areas have dominantly sloped landscapes, while the distribution of flat areas is limited to nearby floodplains of the main river from Andong to the estuary and some sections of major tributaries at the centre of the basin (Figure 1.2 (c)).

This surface relief is controlled by the characteristics of geology in the catchment (KIGAM, 2020) (Figure 1.2 (e)). The high mountains at the northern and western edges of the basin are made of the Precambrian metamorphic rocks, which are resistant to weathering. In these areas, the granite, which was intruded during the Mesozoic era, also dominates the landscape by forming the steep and high mountains and supplies a large amount of sand to the river network. At the centre of the basin, sedimentary rocks from the Mesozoic era are extensively distributed, forming the base of the basin where the intrusion of granite and volcanic rocks created the barrier of the mountains on the east side of the basin.

These morphological and geological features affect the climate system in the catchment (Figure 1.2 (f)). Temperature-wise, the catchment has a big variation between summer and winter and between the north and the south. During winter, the northern region of the study area has below 0 ℃ for the mean temperature, while the south part of the catchment is warmer with a mean temperature of around 0 ℃. In summer, the catchment has a high temperature of over 30 ℃ in the max temperature and two cities Daegu and Miryang are well-known for the summer heatwave. The precipitation pattern in the catchment is contrasting in the season. During summer, the catchment usually receives more than 70 % of annual precipitation, driven by the monsoon and typhoons from late June to early September, while winter is mostly dry with little precipitation. The catchment has also spatial variations in the precipitation (Korea Meteorological Administration, 2012). The south area has relatively high precipitation with an annual mean of over 1300 mm due to their closeness to the coastline, followed by the north region (e.g. Andong and Sangju) with 1200 mm due to the effect of orography. However, the mid-region of the catchment such as Gumi and Daegu is the driest with around 1000–1100 mm per year due to their geographical distances from the sea.

The catchment has the Nakdong river as the main river, originating from the northernmost tip of the catchment to the South Sea with a length of 511 km (Figure 1.2 (a) and (d)). Two large multi-purpose dams directly regulate flow into the main river: the Andong dam (D_ANDO) in the upstream of the main river and the Imha dam (D_IMHA) in the upstream of Banbyeon Stream. The river flows south through populous cities like Andong, Sangju, Gumi, Daegu, and Busan in the floodplains and the lowland. In particular, Daegu, Jinju, and Busan are major cities with a large population of more than a million and account for the substantial urban area. Along the main river, a large industrial complex is situated in Gumi, which is well-known as one of the major water pollution sources in the main river. At the river mouth, the main river is blocked by estuary banks from the South Sea. In the catchment, the mountainous areas are mostly covered by forest and valleys in those areas are heavily cultivated for agriculture (Figure 1.2 (d)). For example, the north and west areas of the catchment are mostly used for orchards and fields, while paddy fields and greenhouse farms are the dominant types of agricultural land use on the floodplain along the main river below Sangju

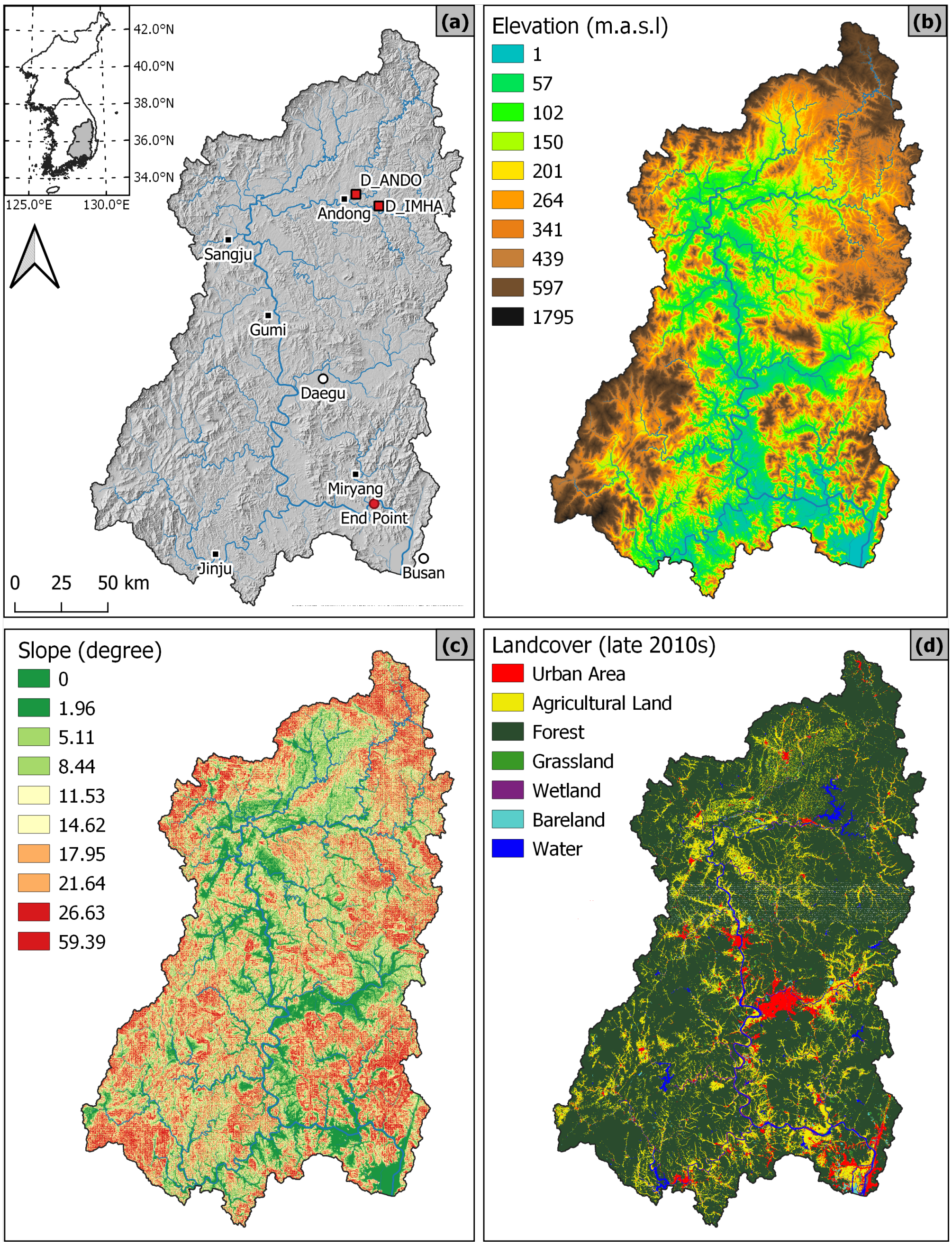


Figure 1.2: Maps of the study area: the Nakdong river and its tributaries (stream order > 6) in the Nakdong catchment (a) Major cities and the rivers, (b) Elevation, (c) Slope, (d) Landcover. These maps were generated in QGIS using data from the Water Resources Management Information System (2019), the National Institute of Ecology (2021), the KIGAM (2020), and the KMA (2020)

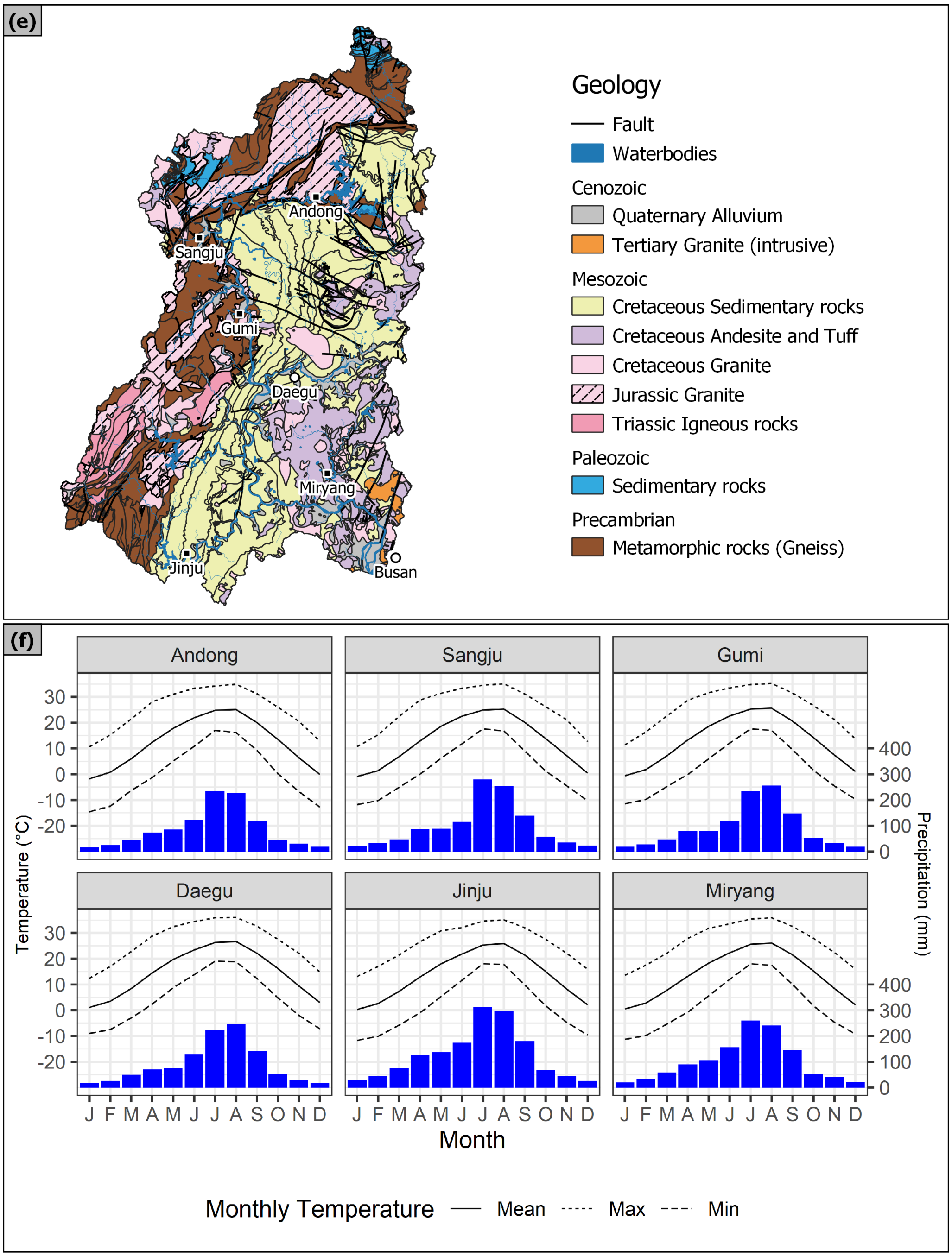


Figure 1.2: (continued)

and the tributaries.

To tackle water pollution in the catchment, the South Korean government make effort such as the enforcement of a stricter criterion on phosphorus concentration in the released water and the installation of sewage treatment plants. Nevertheless, pressure on land use and water and climate change will likely affect catchment processes, water quality and riverine ecosystems in the catchment. For instance, Kim and Choi (2013) report that the runoff in the Keumho catchment in the Nakdong catchment will increase by 17–46 % in 2050 during spring and winter and decrease by 12–20 % during summer and autumn under the scenarios of RCP 4.5 and 8.5. Furthermore, with the scenario of the expansion of the urban area in the catchment on top of the climate change models, the change in runoff will increase by 1–3 % during spring and summer and decrease by 2–7 % during autumn and winter. Similarly, Jang et al. (2015) find that the Imha dam (see the location in Figure 1.2 (a)) catchment will experience an increase in precipitation by -2.1–10.8 % with an increase of the mean temperature by up to 0.8–5.0 °C by the 2080s under the RCP 4.5 and 8.5 scenarios. Therefore, it is important to understand the Nakdong catchment system for future projection and catchment management planning.

In this study, unless stated otherwise, the spatial extent of the main river is restricted to reaches between the two dams upstream and 1 km below the confluence where the Miryang river joins the main river downstream (Endpoint in Figure 1.2 (a)) in order to primarily focus on the impact of the barrages on the riverine ecosystem. The tributaries in the catchment considered for this project are 11 major rivers and streams in the order from upstream of the main river; Banbyeon Stream (BB), Naeseong Stream (NS), the Young River (YG), Byeongseong Stream (BS), We Stream (WE), Gam Stream (GM), the Keumho River (KH), Hoe Stream (HO), the Hwang River (HG), the Nam River (NG), and the Miryang River (MR). They were all selected based on Strahler's stream order (> 6) and standard catchment divisions used by the government for their river management plan (Water Resources

Management Information System, 2019).

1.3 Summary

This chapter has looked at the effort to tackle water pollution in rivers and the importance of understanding biological responses in relation to environmental variables in river health assessment. The study area, the Nakdong catchment in South Korea, contains eight barrages constructed in the Nakdong river as part of the FMRRP in July 2009 to improve water quality and river health. However, the environmental impacts of the barrages on the riverine ecosystem have not been fully examined yet, and remain controversial. Therefore, this thesis will investigate the environmental impacts of the barrages in the Nakdong river and its tributaries by looking at relationships between environmental variables and diatoms in time and space.

Chapter 2

Literature Review

2.1 Introduction

This chapter provides an overview of previous research on the use of diatoms in river health assessment and structures this project with research aim, and questions. The first part reviews diatom-based water quality research in rivers across the world. The second part deals with previous research carried out in the Nakdong catchment on water quality and biological assessment using diatoms in order to find shortcomings and research gaps in the literature. Then, the chapter looks at various factors that can control the distribution of diatoms along with key frameworks to view riverine ecology. This chapter also discusses methodological issues in sampling diatoms in the field and what decisions should be made at the planning stage before conducting fieldwork. Lastly, based on the review, the research aim and questions for this project are addressed at the end of this chapter with the structure of thesis.

2.2 Water quality assessment using diatoms

Diatoms are widely used in water quality studies and have advantages to invertebrates and fish including their important role in ecosystems as primary producers, universal presence in aquatic habitats, utility as indicators with sensitivity to environmental conditions, short lifespan, ease of sampling, and limited mobility (Stevenson et al., 2010; Winter and Duthie, 2000a; Astorga et al., 2012).

Among various approaches to utilising diatoms, for example, biomass, diversity, taxonomic composition, guild indicators, and autecological indices, taxonomic composition is one of the most common methods employed to analyse diatom samples paired with environmental variables measured (Stevenson et al., 2010). In this way, ordination methods such as Principal Component Analysis (PCA), Canonical Correspondence Analysis (CCA) and Non-metric Multidimensional Scaling (NMDS) Analysis are applied to establish relationships between diatom taxa and environmental variables using diatom array data that can be easily created from identification under a microscope and enumeration of taxa. Numerous studies reveal the relationships between various diatom taxa and water quality variables as well as land use types in catchments across the world and identify important indicator species.

For example, Lobo et al. (1995) discovered that diatoms in rivers in Tokyo, Japan were divided into two groups by organic pollution and eutrophication; *Achnanthes minutissima* var. *saprophila*, *Gomphonema parvulum*, and *Navicula minima* were positively correlated with a high level of pollution, while *Synedra inaequaliz*, *Cymbella minuta*, and *Achnanthes convergens* were negatively correlated with pollution. Rott et al. (1998) identified the relationships between diatom taxa and seasonal variables such as date and temperature as well as water quality variables in the Grand river in Canada. In a study on the response of epilithic diatoms to urbanisation influences in streams in Melbourne, Australia, Newall and Walsh (2005) found that *Sellaphora seminulum*, *Rhoicosphenia abbreviata*, *Nitzschia palea*, *Nitzschia inconspicua*, and *Nitzschia frustulum*, *Eolimna minima*, and *Amphora pediculus* were indicative of eutrophic conditions in rivers, positively associated with urbanisation gradients. Walsh and Wepener (2009) attempted to identify diatoms in relation to land use types in South Africa. They revealed that *Amphora pediculus*, *Cocconeis pediculus*, *Navicula gregaria*, *Diatoma vulgaris*, *Aulacoseira granulata*, *Nitzschia frustulum*, and *Nitzschia palea* were frequently found in agricultural areas, while *Eolimna minima* and *Eolimna subminuscula* were strongly related with high nitrogen and high phosphorus levels in urban rivers. In a nationwide study on the distribution of diatoms in South Korea, Hwang et al. (2011) classified diatom into four major groups and analysed their relationships with water quality and land use in the catchments. They revealed that *Achnanthes minutissima*, *Achnanthes convergens*, and *Cymbella tumida* were positively correlated with forest land use and low conductivity, indicating oligotrophic rivers, whereas *Aulacoseira granulata* and *Navicula cryptocephala* showed a strong correlation with urban land use. In addition, *Rhoicosphenia abbreviata* and *Navicula subminuscula* showed a strong correlation with nutrient concentrations. In China, Tan et al. (2014) found several indicator species in rivers in relation to pollutants — *Eolimna subminuscula*, *Navicula cryptotenella*, and *Nitzschia palea* for high TN concentration, *Encyonema minuta* and *Nitzschia dissipata* for high TP concentration, and *Discostella pseudostelligera* for high dissolved organic carbon. Similarly, Chen et al. (2016) identified *Pseudostaurosira brevistriata*, *Staurosira construens* var. *venter*, and *Nitzschia palea* as indicator species representing urban streams in Beijing compared to the reference stream where *Achnanthidium minutissimum* was the dominant species.

The information obtained on the ecological preferences of diatom taxa allowed diatom indices to be developed or modified and they are now widely adapted across the world as a means of biological monitoring for river health assessment: the Trophic Diatom Index (TDI) (Kelly and Whitton, 1995) in the U.K., various diatom indices (Wang et al., 2005) in the U.S.A., the Biological Diatom Index (BDI) (Coste et al., 2009) in France, the Duero Diatom Index (DDI) (Álvarez-Blanco et al., 2013) in Spain, the Polish Index (IO) (Szczepocka et al., 2014) in Poland, the Diatom Assemblage Index to organic pollution (DAIpo) (Watanabe et al., 1986) in Japan. These indices are straightforward and unequivocal based on grade or score and they can facilitate comparisons of river health in time and space.

Despite the indices being successful as a means of biological assessment for river health, most studies using diatoms in river health assessment are heavily restricted to one or a small number of visits per site for sampling on a short time scale and thus are only able to provide a snapshot of the ecological status of rivers at the time of sampling at sites. For example, in most studies, the duration of sampling is usually limited to one year and the frequency of sampling per site ranges from 1 to 6 (see Table A.1). The focus on a short-term approach is generally thought to be associated with the main goal of biological monitoring assessment programmes: to diagnose the ecological status of rivers. These short-term studies may still be valid for a short time period and assess the current status of river health at sites correctly, however, they will have a relatively short life as meaningful research. In addition, it is very likely that more exotic or new taxa are discovered in rivers in response to changes in the geographical distribution of climate zone driven by climate change (IPCC, 2021). As a result, under this short-term approach, the emergence of new taxa may go unnoticed or may lead to an erroneous evaluation of river health until the ecological preferences of new taxa are better known (Lee et al., 2021). The prevalent tendency of a short-term approach in the literature also makes it difficult for environmental engineers to evaluate the efficacy of river management policy and to set a direction for the future. Lastly, short-term studies with limited sampling are more vulnerable to erroneous results in river health assessment because samples may not be representative due to different timing, local issues, or errors in sampling methods which will be discussed later. Kelly et al. (2009) expressed their concerns over the trend in the application of diatoms in river health assessment and emphasised the importance of long-term studies with as many visits to sampling sites as possible.

In the literature, undoubtedly there is a lack of long-term research on the application of diatoms with a tendency for a short-term approach using indices (see Table A.1). Consequently, changes in the species composition of diatom assemblages on a long-term scale have been left ignored. In addition, short-term studies tend to explain diatoms in relation to water quality variables, and this has left the effects of seasonality and temporal changes in diatom assemblages overlooked in river health assessment. Ultimately, these tendencies will undermine the value of diatoms as biological indicators. Therefore, a long-term approach using multiple years of consecutive diatom monitoring data with a high frequency of sampling per year is needed to establish the relationships between diatoms and various environmental variables.

To date, only a few notable studies deal with multiple years of diatom records or understand the responses of diatoms to other environmental variables. Snell et al. (2014) looked at the responses of diatom assemblages using the consecutive monthly monitoring data for two years in headwater streams in the U.K. and concluded that diatom communities best reflect stream discharge conditions and Total Phosphorus concentrations during the preceding 21 days. Later, Snell et al. (2019) constructed the recurring seasonal baseline from a ratio of certain diatom taxa using monthly monitoring of diatom for six years in low order streams in England. Similarly, Zou et al. (2018) demonstrated seasonal changes in diatom assemblages in a Chinese lake over two years. In France, Alric et al. (2021) made the first attempt to evaluate the ecological quality of rivers on a national scale over the last two decades by integrating three different biological assessments, including diatoms. Regarding other environmental variables, Krajenbrink et al. (2019) revealed a different response from diatom assemblages in sites affected by river impoundment in comparison to unregulated sites. Klamt et al. (2020) and Peszek et al. (2021) revealed the response of diatoms to drought in rivers in Poland and in a Chinese lake, respectively.

2.3 Review of previous studies on the Nakdong river

2.3.1 Water quality research

Water quality research in South Korea has received a large amount of attention since the phenol leakage incidents in the Nakdong river in 1991. Most studies were primarily focused on analysing the spatio-temporal pattern of water quality variables or tracking down trends of the variables such as pollutants based on measurements of the physical and chemical properties of water in the Nakdong river and its tributaries (Chun et al., 1999; Jung et al., 2016).

In the Nakdong river, total nitrogen, total phosphorus, and organic pollutants (total organic carbon and chemical oxygen demand) are major components in explaining water pollution and a high level of pollution is mostly observed in monitoring stations going through the centre of cities such as Gumi and Daegu (see the locations in Figure 1.2) or in the midstream and downstream of the main river (Jung et al., 2016), primarily due to the large populations of these cites and effluents from industrial and residential areas (see land cover map in Figure 1.2). Within the catchment, the Keumho river (KH), flowing through Daegu to join the main river, is one of the major tributaries responsible for pollution to the main river and is urged to be prioritised for the improvement of water quality in a study of water quality assessment of 28 tributaries (Jung et al., 2020).

In regards to the construction of the barrages in the main river, Hwang et al. (2013) analysed the impact of the construction on water quality at one site in the main river by monitoring various water quality variables daily during four months of the construction period and concluded that the construction did not directly affect water quality except suspended solids and turbidity. Meanwhile, in a study on the effects of the barrage construction on phytoplankton assemblages and water quality in the study area, Lee et al. (2018) found an increase in harmful algae during summer and autumn compared to the pre-construction period due to changes in hydrological factors. Yu et al. (2014) revealed the dominance of cyanobacteria during high water temperatures in summer, which is strongly correlated with water temperature and partially with daily solar radiation during the summer in the Nakdong river after the construction of the barrages. Yu et al. (2015) demonstrated that the increase in the retention time of water significantly contributed to the increase in chlorophyll-a concentration with alkaline water on the surface in stratified water in lab experiments. This result explains the increase in algal blooms in the main river after the construction of the barrages. Jung et al. (2014) recorded similar results demonstrating that phytoplankton abundance is influenced much more by hydrodynamics than nutrients in the Han river in Korea.

Figure 2.1 presents the expected hydrological changes and the expected interactions in physical, chemical, and biological processes in the Nakdong river after the construction of the barrages based on the results of the previous studies (Hwang et al., 2013; Yu et al., 2014; Yu et al., 2015; Lee et al., 2018). However, the results of the previous studies are entirely reliant on short-term observations of

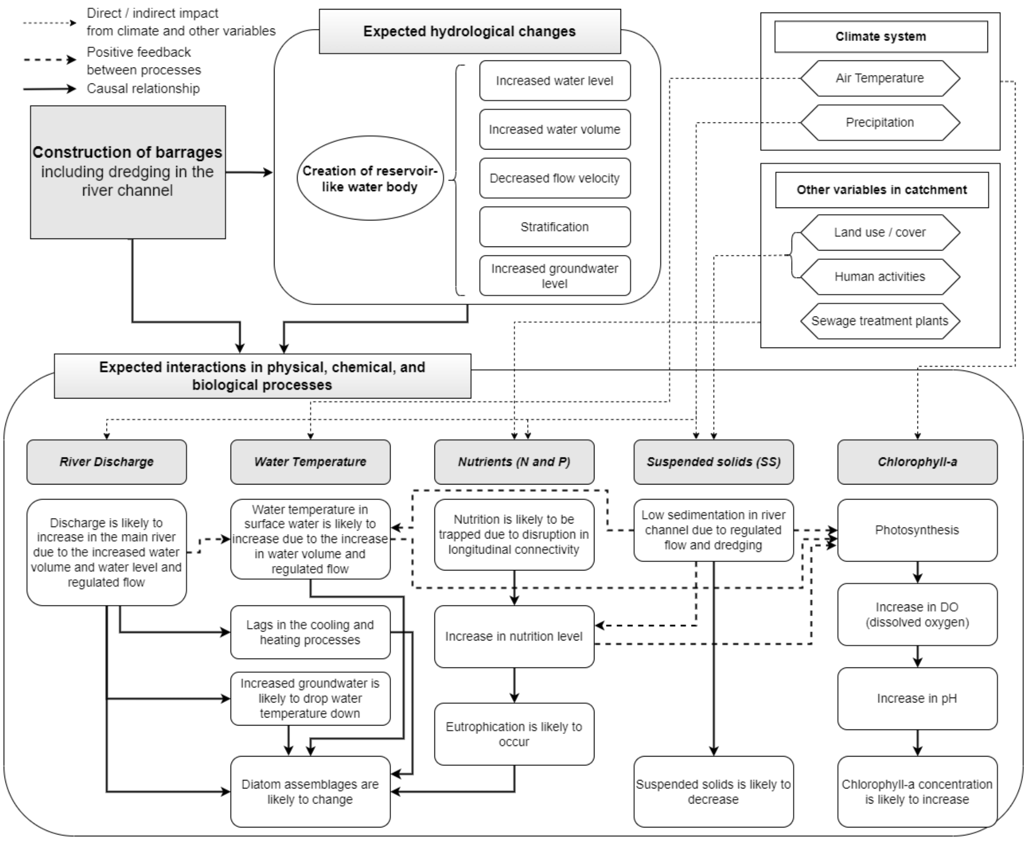


Figure 2.1: Expected hydrological changes and interactions in physical, chemical, and biological processes after the construction of the barrages

water quality data which could be biased due to uncontrolled variables. (e.g. three years, Yu et al., 2014; Yu et al., 2015; four months of monitoring, Hwang et al., 2013; comparisons between 2006–2007 and 2013, Lee et al., 2018). Therefore, this short-term approach is not ideal for fully assessing the impact of the barrages on water quality and ecosystems in the Nakdong river. The true picture can only be revealed when long-term observations are undertaken. The understanding of spatio-temporal changes in water quality in the Nakdong river will be a foundation to understand the responses of aquatic ecosystems including diatoms to environmental changes such as the construction of the barrages.

2.3.2 River health assessment using diatoms in South Korea

The development of integrated water environment assessment programmes in the mid-2000s brought a paradigm shift to river management planning in South Korea (Choi, 2006; Hwang, 2007). Since 2009, as one of the biological assessments, epilithic diatoms have been routinely sampled twice a year in spring (May to June) and autumn (September to October) across the river network to assess the ecological status of river health along with benthic macroinvertebrates and fish (Ministry of Environment, 2017).

Biological assessment using epilithic diatoms in South Korea is evaluated using the modified Trophic Diatom Index (mTDI)*. The index is based on the Trophic Diatom Index (TDI) developed by Kelly and Whitton (1995) and modified by deducting a score from 100 to have a reversed value, ranging from 0 in a poor state to 100 in an excellent state (see Equation 2.1) (Noh et al., 2009; Watershed and Ecology Research Team, 2015).

$$mTDI_{\text{site}} = 100 - \{(WMS \times 25) - 25\} \tag{2.1}$$

where WMS is weighted mean sensitivity and is obtained through equation 2.2.

$$WMS_{\text{site}} = \frac{\sum AiSiVi}{\sum AiVi} \tag{2.2}$$

where Ai is abundance of ith species in sample (%), Si is pollution sensitivity of ith species ($1 \leq S \leq 5$) depending on sensitivity determined by PO_4-P concentration (mg/L), and and Vi is indicator value of ith species ($1 \leq V \leq 3$) determined by probability of Monte Carlo Test. S is large when the concentration is high, and V is large when it has a probability < 0.01 (see Watershed and Ecology Research Team 2015, p.25 for S and V for species).

*The index used in Korea is often named "TDI" in Korean literature despite it being modified. However, for the sake of clarification, this study uses the term "mTDI" for the one used by the National Institute of Environmental Research in South Korea

The mTDI scores are divided into five grades — Excellent (100–90), good (90–70), satisfactory (70–50), bad (50–30), and very bad (30–0) — and *Achnanthes minutissima, A. convergens* and *Cymbella minuta* are abundant in a good state of water quality, while *Gomphonema parvulum* and *Nitzschia palea* are indicators for a very bad state in Korea (Watershed and Ecology Research Team, 2015). Some results obtained from the annual monitoring programmes have been published in Korean scientific journals (e.g. Yoon et al. 2010; Kim et al. 2012; Kim et al. 2017; Choi et al. 2019 in Table A.1).

However, since each survey in spring and autumn is limited to just one visit per site, these publications merely exhibited a snapshot of the ecological status in rivers at the time of sampling in relation to water quality variables measured. This tendency is generally found in annual reports for this national river health monitoring programme (Lee and Hwang, 2008; Lee, 2009; Hwang, 2010; Hwang, 2011; Hwang, 2012; Hwang, 2013; Hwang, 2014; Hwang, 2015; Hwang, 2016; Hwang, 2017; Hwang, 2018). These reports present that the index showed high correlations with water quality variables, especially nutrients. However, they failed to provide any scientific reasons for a rise or fall in the mTDI score at sites compared to the results of previous years but speculated about the possibility of alternation to physical habitats or the degradation of water quality in rivers.

The degradation of the mTDI scores is often attributed to the influence of land use and land cover in the Nakdong catchment. Lee et al. (2021) found poor results in the diatom index in the mid- and downstream of the main river using samples collected in 2014 and concluded that they resulted from effluents from the Gumi Industrial complex and Daegu metropolitan area (see their locations in Figure 1.2. In the upper region of the Nakdong catchment, Choi et al. (2019) presented low scores of the mTDI in Naeseong Stream (NS) and We Stream (WE) using samples taken in 2016 and explained that they were influenced by agriculture and effluents from cities and industrial areas in the catchments.

In the Korean literature, little attempt is made at regional or national scales to examine the distribution of diatoms and understand their relationships with various environmental variables. Indeed, only one large-scale study based on the national monitoring programme exists. Hwang et al. (2011) analysed the distribution of diatoms in Korean streams and rivers in relation to water quality and land use by using the result of the 2009 national monitoring programme. This study was the first attempt made at a large spatial scale to understand the relationships between diatoms and environmental variables in rivers, and it is meaningful as an archive to present the ecological status of rivers in Korea before the construction of the barrages began in four major rivers (including the Nakdong river). However, the result of this study may no longer be valid after the construction of the barrages in rivers, and follow-up research at national or regional scales has yet to be carried out. Recently, Lee et al. (2021) looked at the responses of epilithic diatoms in the Nakdong river using samples taken in 2014 (the post-construction period). However, they focused on understanding the distribution of diatoms in relation to water quality variables after the construction of the barrages and did not compare their findings with samples from the pre-construction period.

Since 2009, a significant amount of the government diatom datasets have been amassed through the river health monitoring programme. Although, several studies make use of the government datasets and report that the mTDI displays a good performance of ecological evaluation in Korean rivers (Kim et al., 2017; Choi et al., 2019; Lee et al., 2021). However, those government datasets have been left without being used in the best fashion. In the Korean literature, short-term approaches are dominant with the duration of one year and one to four samples per year in the frequency (see Table A.1) and long-term approaches to look at temporal aspects of river health assessment are often neglected. No research has been carried out to compare the results of the index over time with changes in water quality or examine the tendency of the ecological status of rivers as a whole. In addition, there has been little interest in exercising species composition in diatom assemblages in those datasets. As a result, the government diatom datasets are not best utilised in addressing current environmental problems in rivers. Therefore, it is beneficial to make the best use of the government diatom datasets by looking at changes in the species composition in diatom assemblages in time and space over time. This could be a key to enhancing our understanding of the responses of diatoms to environmental changes, such as the construction of the barrages. Ultimately, these understandings will help environmental engineers cope with environmental problems better from a holistic perspective.

2.4 The evolution of riverine ecology and diatoms

The understanding of riverine ecology has been a common interest shared among many river ecologists and fluvial geomorphologists (Poole, 2002) and it is increasingly important to understand riverine ecosystems in terms of river management and biodiversity due to the degradation of water quality and ecosystem health in rivers around the world. As a result, numerous frameworks have been proposed to conceptualise riverine ecosystems and understand the complexity of these ecosystems.

One such notable framework is the River Continuum Concept (RCC) (Vannote et al., 1980). This concept depicts the river system as having continuous gradients of physical conditions including width, depth, velocity, flow volume, and water temperature from headwater to river mouth and those gradients ultimately determine the distribution of biological communities in the river. A longitudinal linkage in the river is important in this concept and the characteristics of lotic communities in the river are different according to stream size based on stream order.

Contrary to this view that sees a river as a continuum, the concept of patch from landscape ecology was introduced to the discipline and led to the development of models that view a river as an array of discontinuum. Pringle et al. (1988) and Townsend (1989) applied patch dynamics concepts to lotic systems and concluded that these perspectives provide a unifying concept to understand stream ecology.

These patch concepts were adopted from other disciplines such as ecology, hydrology, and geomorphology and this effort brought fruition to holistic and interdisciplinary frameworks across the disciplines. Thoms, Parsons et al. (2002) proposed the Eco-geomorphology concept as an interdisciplinary framework to help understand riverine ecosystems better through the synchronisation of spatial scales across ecology, hydrology, and geomorphology. Similarly, Benda et al. (2004) attempted to understand the nonuniform distribution of riverine habitats in relation to river network elements such as basin size and shape, drainage density, and network geometry, which ultimately regulate physical habitats in time and space. Thorp et al. (2006) proposed the Riverine Ecosystem Synthesis (RES) by combining the continuum and discontinuum perspectives to understand riverine ecosystems. This framework views rivers as downstream arrays of large hydrogeomorphic patches formed by catchment geomorphology and climate. These hydrogeomorphic patches are different in physical and chemical conditions and create the Functional Process Zones (FPZ) in rivers where different types of lotic ecosystems are defined.

These views see rivers as having continuous gradients or arrays of patches and are scale-dependent at spatial ranges from watershed to habitat. The application of these frameworks at an appropriate scale provides a great platform to simplify complex riverine ecosystems and enhance our understanding of stream ecology.

In understanding riverine ecosystems, it is important to take a holistic perspective, and these frameworks should be understood in consideration of four dimensions — longitudinal, lateral, vertical, and temporal aspects — to understand ever-changing riverine ecosystems. Rivers are connected in longitudinal, lateral, and vertical dimensions within the river channel and the watershed (Wohl, 2017). Therefore, lotic ecosystems interact with patterns and processes in rivers across space and time (Ward, 1989). This heterogeneity in space and time may change in response to physical disturbances such as storms, floods, droughts, or anthropogenic activities and as a result, a different structure of heterogeneity in space can form in rivers. In particular, the temporal aspect is important as it displays how riverine ecosystems respond to those physical disturbances.

In the Nakdong catchment, the Nakdong river and its tributaries would have experienced significant environmental changes due to the construction of the barrages. Therefore, understanding these conceptual frameworks in four dimensions is vital as groundwork to examine the responses of riverine ecosystems to the construction of the barrages in the Nakdong river and its tributaries.

2.4.1 What controls the distribution of diatoms?

The distribution of diatoms was traditionally considered cosmopolitan. The traditional perspective (Finlay, 2002) believed that diatoms disperse freely and flourish once the environment requirements such as physical parameters (e.g. temperature and salinity) and biological parameters (e.g. prey and predators) are met. However, this view has been challenged by recent research (Soininen, 2007; Astorga et al., 2012; Soininen et al., 2016) and there has been an ongoing debate (see Martiny et al. 2006) on whether or not these unicellular organisms show biogeographic patterns at a global scale. Soininen (2007) found that 20–30 % of variation in the distribution of diatoms was explained by pure spatial factors at intermediate and regional scales. This suggests that diatoms have clear biogeographical patterns. At a global scale, Soininen et al. (2016) revealed significant differences in diatom species composition among the Antilles, France, Finland, New Zealand, and the United States and concluded that diatom species distributions are under strong microevolutionary constraints. In a study in Finland where distance decay of similarity in freshwater communities was examined at a 1000 km of the geographical distance scale, Astorga et al. (2012) discovered that the similarity of diatom communities decreases with increasing geographic distance.

Nevertheless, the distribution of diatoms at a global scale is not likely to affect their distribution in South Korea significantly because the country has a geographical gradient of around 500 km in the mid-latitude zone and the distribution of diatoms at a regional scale may be controlled more by local environmental variables that create more gradients. In addition, a study on the nationwide distribution of diatoms in South Korea (Hwang et al., 2011, Figure 6 in p.S26) revealed that major indicator species were found across the nation in different proportions.

At different spatial scales from a region to a substratum, many attempts have been made to explain the distribution of diatoms in rivers in relation to indirect or direct controls of various biotic and abiotic factors. Stevenson (1997) conceptualised the hierarchical framework of determinants on heterogeneity of the benthic algal community at different spatial scales (Figure 2.2).

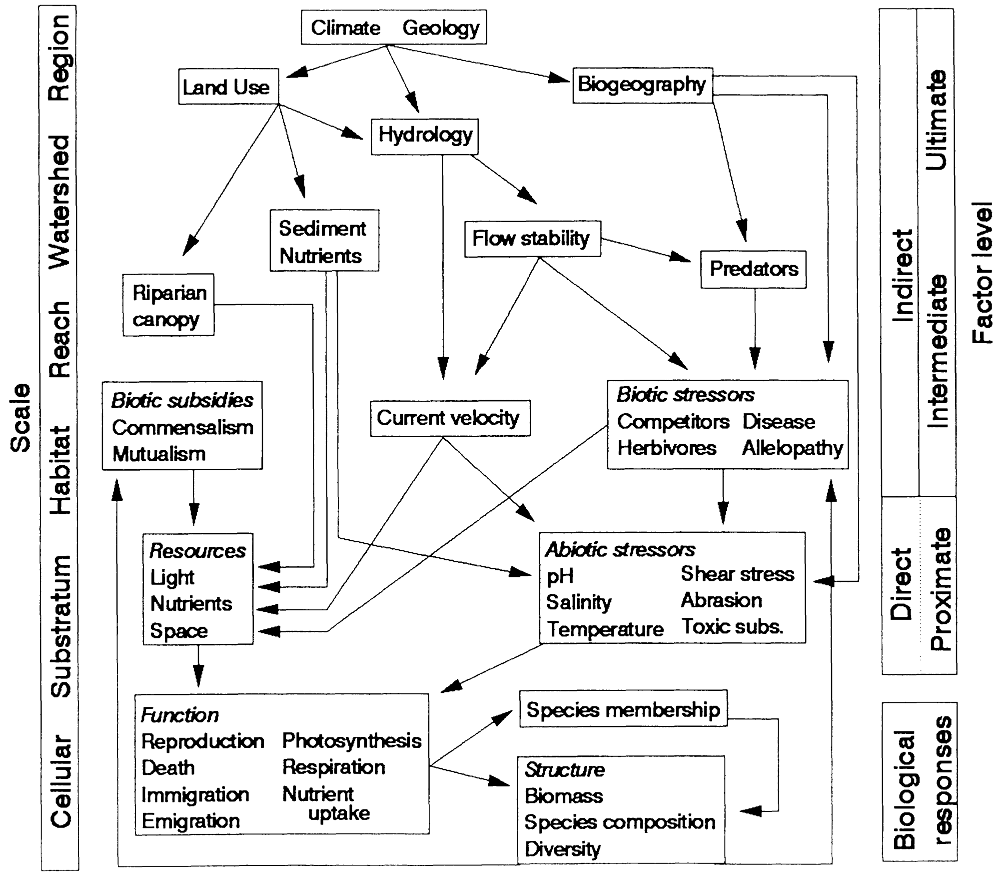


Figure 2.2: A conceptual framework of scale-dependent determinants of the benthic algal heterogeneity (Stevenson, 1997)

From regional to watershed scales, factors like climate, geology, land use, and hydrology may determine the distribution of diatoms in rivers by directly and indirectly affecting physical habitats. For example, Pan et al. (1999) revealed that six diatom groups in Mid-Atlantic streams in the United States were discriminated best by a watershed land cover (forest), riparian conditions, and water temperature. Biggs (1995) examined the contribution of flood disturbance, catchment geology, and land use to the habitat template of periphyton in streams in New Zealand and concluded that the frequency of floods, the proportion of high-intensity agricultural land use in the catchment, and the proportion of alkaline rocks were the most significant factors in explaining variation in chlorophyll-a concentration. Newall and Walsh (2005) revealed that basin and urban variables were the most effective in explaining the patterns in diatom assemblages in streams in Melbourne, Australia when they were combined with drainage connection, elevation, and longitude.

Hydrology in rivers and its changes such as flow variability may play a pivotal role in determining the distribution of diatoms. Bunn and Arthington (2002) conceptualised basic principles for successive ecological changes to altered flow regime. Molloy (1992) looked at the longitudinal distribution of diatoms from headwater to downstream environments in the Kentucky river in the United States. He discovered relationships between morphological growth forms of diatoms and stream size, and the contribution of currents to diatom taxa; erect and stalked taxa were more commonly associated with headwater environments, while filamentous and centric diatoms occurred with greater frequency downstream. In addition, the responses of diatoms to river impoundment in England (Krajenbrink et al., 2019), to drought in Polish rivers (Peszek et al., 2021), and in two Chinese lakes (Klamt et al., 2020) are evidence of the role of hydrology.

From habitat to substratum scales, biotic factors like competitors and herbivores or abiotic factors such as pH, salinity, temperature, light, and nutrients may affect benthic algal assemblages. Lange et al. (2011) revealed that light had the strongest influence on diatom community structure, whilst nutrients had a medium effect, and grazing was relatively weak. Peszek et al. (2021) explained the significant difference in seasonal variability of diatom assemblages between low order and high order stream sites in Polish rivers in relation to light availability, affected by riparian vegetation.

In reality, it is difficult to distinguish one factor from another in explaining the distribution of diatoms because riverine ecosystems are complex features that are simultaneously influenced by many variables at a range of scales (Thoms, 2006). In addition, these factors are spatially and temporally scale-dependent. As a result, the effect of some factors may be obscured by others. Therefore, it is important to understand the distribution of diatoms at appropriate scales depending on research objectives.

2.4.2 Methodological issues in sampling diatoms

A lot of effort has been made to improve and standardise diatom sampling methods across the world (Round, 1991; Kelly et al., 1998; Lazorchak et al., 1998; Lee et al., 2016). In general, there are some preferred methods in the literature, nevertheless, sampling methods such as types of diatoms, types of substrate, the number of samples, and timing and frequency of sampling still depend on researchers' judgement or research objectives. Thus, sampling methods should be carefully assessed during the planning stage because selection errors in sampling methods could lead to erroneous results or different sampling methods make it difficult to compare the results with others fairly. In this section, three aspects of sampling methods are reviewed with implications.

2.4.2.1 What type of diatoms and substrata to sample?

Diatoms are largely divided into five types by habitats: planktonic/phytoplanktonic (floating), epiphytic (attached to macrophyte), epilithic (attached to rock), epipsamic (attached to sand), and epipelic diatoms (attached to silt) (Round, 1991). These habitats are usually unevenly present in rivers in time and space and this often makes the selection of diatom type and substrate challenging. Therefore, at the large spatial scale, the selection of diatom type and substrate should be carefully considered depending on the study objectives and the availability of habitats in the study area.

Despite the ease of sampling, in practice, planktonic diatoms are mostly avoided in the literature because of strong seasonality in some common species and possible contamination by species from upstream (Round, 1991). It is worth noting that Lane et al. (2007) reported the presence of a substantial proportion of rare diatom taxa in a phytoplanktonic diatom community compared to periphyton diatom communities collected from rocks, debris, and sediments in the littoral zone in four American rivers. This suggests that there is a high risk of getting biased samples when planktonic diatoms are sampled and they should be avoided unless it is the research objective.

Two major sampling methods exist for benthic diatoms (Weilhoefer and Pan, 2007); the reach-scale method that collects a predetermined number of samples throughout the reach on different substrata (e.g. rock, macrophyte, sand) and in different habitats (e.g. riffle, pool); and the targeted-habitat method that only gathers samples from a single and dominant habitat type within the reach. The former has the advantage that a mixture of diverse diatoms in different habitats can be captured, but differences due to environmental conditions might be hidden by the diversity in habitats or substrata. In comparison, the latter should reflect a difference caused by environmental conditions more accurately since it is free from noise that may be introduced by the inclusion of different habitats or substrata (Stevenson and Bahls, 1999).

The assessment on the performances of two sampling methods remains contradictory in various studies. Winter and Duthie (2000b) demonstrated that there was no apparent benefit to discretely sampling three different habitats (rock, silt, and macrophyte), but went on to recommend that the best relationship can be obtained from using the sum of all of them. However, Weilhoefer and Pan (2007) found out that there was no significant difference in diatom assemblages between the reach-scale method and the targeted-habitat method.

For the targeted-habitat method, rock samples have been widely used due to their common availability in rivers as well as for the relative ease of sampling in comparison to macrophytes or sediments (Round, 1991; Kelly et al., 1998). The use of rock samples is advocated by the results that substrate types do not cause any significant difference in diatom community structure (Rott et al., 1998; Winter and Duthie, 2000b) or do not affect the overall assessment of river health (Richards et al., 2020).

On the contrary, some studies advocate the use of other substrata instead of rock samples. Besse-Lototskaya et al. (2006) suggested that better results can be obtained from using macrophytes for river bio-monitoring. Likewise, Vidaković et al. (2018) reported less variation in the epiphytic diatom community compared to the epilithic community in a Russian river and highlighted the important role of epiphytic diatoms to supplement epilithic diatoms in river bio-monitoring. Kahlert and Savatijević Rašić (2015) showed that diatom assemblages between macrophytes and rock samples were significantly different at a 1 m spatial scale (within 1 m) and this difference in diatom assemblages may affect the results of indices.

In the absence of appropriate natural substrata to sample in rivers, artificial materials or barriers such as pillars or quays which are submerged under the water can be sampled instead (Kelly et al., 1998). Richards et al. (2020) advocated the use of rope over rocks or sediments as a substrate based on their findings that ropes do not emit any nutrients that subsequently might affect the growth of diatoms. However, the use of artificial materials sometimes requires more time and effort as they have to be set up and collected later, necessitating at least two trips to the sites. They are also exposed to a great risk of potential vandalism (Stevenson et al., 2010).

To sum up, the type of substrata to sample is entirely reliant on the study objectives and the availability of substrata in the targeted river (Stevenson et al., 2010). Nevertheless, within the discipline as a whole, rock samples have remained the most popular as a substrate and the recommendations made by Kelly et al. (1998)

serve as general guidelines for the practice. According to their advice, rock samples that are unlikely to move under normal hydrological conditions should be randomly collected at different parts of the reach and cobbles are much preferred over boulders purely because of the convenience of handling. Alternatively, pebbles can be used where cobbles are not available. They also supported the existing practice of taking at least five cobbles, but suggested that the number should be increased to at least ten when pebbles are to be sampled.

2.4.2.2 How important is variability in diatom assemblages at the reach scale?

In the targeted habitat method, rock samples have been widely used for biological assessment in rivers and in the process, a composite sampling method is widely used. A composite sample is made from merging multiple rock samples or a single rock sample randomly taken within a reach (Stevenson et al., 2010; Kelly et al., 2009). The rationale for this method is that samples within a reach are representative and random multiple samples of one habitat type must be similar to each other, mainly controlled by the same environmental variables.

However, despite the prevalent use of a composite sample from multiple random samples at a reach scale, variability in diatom assemblages among those multiple random samples at a reach scale has been overlooked in the literature with only three notable studies within a reach scale. Passy (2001) investigated the spatial variability of diatoms at a reach scale (4 m × 5 m) in a stream where rock samples were collected at intervals of 0.5 m. Passy (2001) revealed the different responses of two major diatom species — *Achnanthes minutissima* and *Fragilaria capucina* — to velocity in the stream; *A. minutissima* increased in relative abundance with increasing velocity, while *F. capucina* decreased. This result demonstrates that epilithic diatoms can differently respond to physical variables within a reach scale. Snell (2014) investigated the patterns in benthic diatom communities in the riffle-pool-riffle-pool sequence in headwater streams in the U.K and discovered different characteristics; the pool has more diversity in diatom assemblages composition with higher species richness and diversity, while the riffle has more productivity and variability in diatom assemblages. Kahlert and Savatijević Rašić (2015) analysed the variability of diatoms at centimetre to decimetre scales at one site in a mesotrophic Swedish river using two different substrata stones and plants. They discovered no significant difference in variability of diatom assemblages between samples from at a centimetre scale and a decimetre scale and concluded that small scale variation is not an issue for river health assessment. However, this example only included samples at one site in the river, and variability in environmental variables at centimetre to decimetre scales might not be as significant as at a reach scale.

In conclusion, although the selection of a single dominant habitat type (mostly rocks) is preferred on the basis that diatom assemblages on a specific substrate are similar and thus habitat variability can be minimised, multiple random samples at a reach scale may still contain some degree of the heterogeneity in diatom assemblages as demonstrated in the work of Passy (2001). Without understanding variability in diatom assemblages from multiple random samples at a reach scale, the use of a composite sample may lead to erroneous results at the sites, and subsequently, undermine the value of diatoms as a biological indicator. Therefore, it is necessary to examine what variability at a reach scale is expected to happen from random sampling and to understand what variability in diatom assemblages means in terms of water quality assessment. Ultimately, the examination of replicability in diatom assemblages at a reach scale will provide a good threshold for diatoms to be considered as replicate samples.

2.4.2.3 When and how often to sample?

The timing and frequency of sampling in rivers should be carefully considered as some taxa are seasonal and sensitive to changes in temperature, hydrological conditions, and physical disturbances. In general, sampling is generally recommended at the timing of low flow or expected concentration of pollution (Kelly et al., 1998). For water quality assessment in rivers, diatoms are usually sampled in spring and autumn when hydrological conditions are easy to access without being affected by extreme discharge (e.g. Korea, Watershed and Ecology Research Team 2015; the U.K., Kelly et al. 2008) but sampling is purely objective dependent. In a one-year diatom monitoring project in a German lowland river, Wu et al. (2016) revealed that the structure of planktonic diatom assemblages changed as quickly as within a day during the wet season, and as slow as more than 30 days during the dry season. They noticed that storms and precipitation contribute to the fast-changing diatoms in the river during the wet season by supplying nutrients to the river from the catchment and concluded that sampling frequency should be increased during the wet season. This result demonstrates the significant effect of hydrological conditions on diatoms and it is in line with the recommendation (Kelly et al., 1998).

In terms of sampling diatoms in rivers for water quality assessment, the effect of temperature on diatom assemblages is not importantly considered. Kelly et al. (1995) demonstrated that samples taken at four different times of the year in rivers in England and Scotland showed no significant influence of season on any of the indices. As a result, water quality assessment using diatom indices is heavily reliant on one or a few visits per site during spring and autumn. However, dependence on the limited number of sampling per site may induce erroneous results in river health assessment because of insufficient samples. Kelly et al. (2009) raised concerns over the misclassification of ecological status based on a single sampling and recommended the use of several samples collected at a site spread over a year. In South Korea, one visit per site is usually made for diatom sampling during each of the two month periods in spring and autumn (Watershed and Ecology Research Team, 2015). However, there is a chance of variability in diatom assemblages within the survey period. Moreover, some diatom taxa are known to have seasonality (Round, 1991) but little effort is made in the literature to understand the responses of diatoms to temperature and seasonal changes in rivers except for two notable studies

(Snell et al., 2014; Snell et al., 2019). Therefore, variability in diatom assemblages during the South Korean national monitoring period needs to be examined as an example to enhance our understanding more.

2.4.2.4 Implications for approaches to sampling

It is important to determine what type of diatoms and habitats should be sampled and when and how often sampling should be conducted at the planning stage of a research project. These decisions should be made with the research objectives in mind in consideration of existing datasets and previous literature in the targeted study area or nearby. The consideration of the methods at the planning stage will subsequently ensure reliable results with little chance of errors.

In this project, epilithic diatoms from multiple rock samples in rivers are used in the same manner that the Government Diatoms (GOVD) dataset has been established for the sake of convenience in comparison. Timing and frequency for sampling are also determined depending on each chapter's objective and separately stated in the Methods section of each chapter.

2.5 Summary

This chapter has reviewed the literature on diatom-based water quality assessment from around the world. The focus then switched to South Korea and looked at the trends of water quality research and the national river health assessment using diatoms in South Korea. In addition, the major frameworks in riverine ecology, as well as various determinant factors for diatoms in rivers, were discussed to better understand the distribution of diatom communities. Finally, methodological issues in sampling diatoms were scrutinised and appropriate diatoms sampling methods for this project were introduced. The review of previous literature identified research gaps in the literature and in South Korea. They are used to formulate research questions to make this research project more reliable. Before looking at the research aim and questions, key points found in this chapter are summarised as follows.

1. Diatoms are widely used as indicators in water quality and river health assessments across the world. Understanding relationships between diatoms and environmental variables and the identification of indicator species led to the development of diatom indices to assess the ecological status of rivers. Despite the indices being successful as a means of river health assessment, most studies in the literature are heavily restricted to one or a small number of visits per site on a short time scale and thus are only able to provide a snapshot of the ecological status of rivers at the time of sampling at the sites. This practice also makes it difficult for environmental engineers to evaluate the long-term efficacy of river management policy. In addition, it is more likely that exotic or new taxa are discovered, driven by the effects of climate change. As a result, under this short-term approach, new taxa may complicate river health assessment. In contrast, there is a lack of long-term research on the application of diatoms in river health assessment amid a tendency for a short-term approach using indices. In Korea, the species composition of diatom assemblages has been left, receiving little attention. Also, due to a lack of long-term studies, temporal changes in diatom assemblages are poorly understood. Therefore, it is necessary to take a long-term approach using species composition for a more holistic river health assessment.

2. Water quality studies in South Korea reveal that total nitrogen, total phosphorus, and organic pollutants are major components in explaining the variance in water quality in the Nakdong river. Hence, a high level of pollution is generally reported in the mid- and downstream of the main river affected by cities with high population density and effluents from industrial and residential areas. In regards to the construction of the barrages in the main river, some studies were carried out to assess the impact on water quality,

phytoplankton assemblages, the dominance of cyanobacteria, and the retention time of water. They reveal the negative impacts of the barrages, however, those studies were heavily dependent on short-term observations in the Nakdong river and they could be easily biased due to uncontrolled variables. Therefore, the true picture of the impacts of the barrages on water quality and riverine ecosystems will be better revealed in a long-term approach.

3. River health assessment in the Nakdong river is regularly conducted by applying the mTDI (modified Trophic Diatom Index) to epilithic diatoms collected once during spring and autumn since 2009. The annual monitoring programme reports speculated the degradation in the mTDI result at the sites would be related to the deterioration of water quality or alternation to physical habitats, but failed to provide a firm reason for the deterioration in river health. Also, the decline in the mTDI results tended to be linked with the impacts of land use in the catchment. River health assessment in South Korea is concentrated on the use of the mTDI in a short-term approach, while the species composition of diatom assemblages has been left ignored and the assessment lacks a temporal perspective. Moreover, little effort is made at a large spatial scale to utilise diatoms in understanding the responses of the riverine ecosystem to environmental changes such as the construction of the barrages, when the Government Diatoms (GOVD) datasets are amassed through the annual river health monitoring programme. Therefore, to use the GOVD more effectively, the species composition of diatom assemblages should be examined in a long-term approach and this will help assess the impacts of the barrages on water quality and riverine ecosystems.

4. Rivers are viewed as continua having continuous gradients of physical variables, or discontinua consisting of patches. These perspectives created several frameworks to explain riverine ecosystems such as the River Continuum Concept (RCC), patch dynamics concepts, and the Riverine Ecosystem Synthesis (RES). These frameworks provide a basis to understand complex river-

ine ecosystems. This understanding should always be sought in four dimensions: longitudinal, lateral, vertical, and temporal dimensions. The heterogeneity of diatoms in rivers is influenced directly and indirectly by various factors such as climate, geology, land use, hydrology, riparian vegetation, and light at different spatial scales from region to substratum. These factors are scale-dependent and may be obscured by others as they simultaneously interact with each other in riverine ecosystems. Therefore, it is important to understand the distribution of diatoms at an appropriate scale depending on research objectives.

5. In sampling diatoms in rivers, types of diatoms and substrata, the timing, and frequency of sampling should be decided ahead of fieldwork to minimise any errors as well as to achieve research aims. In water quality assessment, epilithic diatoms, attached to rocks, are generally preferred because of their common availability in rivers as well as the relative ease of sampling. Sampling is recommended at low and stable hydrological conditions or the expected concentration of pollution. Most studies rely on one or a small number of visits per site for sampling, however, sampling frequency should be increased to conduct more accurate and holistic river health assessments. Overall, no absolute sampling method exists but it should be carefully determined in consideration of research objectives, previous studies in the study area, and existing diatom datasets.

6. Two research gaps have been identified. First, variability in diatom assemblages within the survey period needs to be examined to see relationships between diatoms and environmental variables and how they can be used for better river health assessment. Currently, diatom sampling in South Korea is usually carried out once during the two-month period in spring and autumn throughout the year and one sampling per site within the period may be influenced by variability in diatom assemblages. Second, when it comes to making diatom samples, a composite sample made from multiple random

rock samples or a single sample collected at a reach scale is widely used in the belief that multiple samples by one habitat type within a reach are similar to each other and representative. However, variability in diatom assemblages within a reach scale may arise and cause errors in water quality assessment. Therefore, it is necessary to examine variability in diatom assemblages at a reach scale and test whether diatom assemblages are reproducible from random sampling or not.

2.6 Research aim and questions

Based on the literature review and the identified gaps, the research aim and questions for this project are addressed below.

2.6.1 Research aim

The aim of the project is to understand the relations between water quality and diatom assemblages in relation to the construction of eight barrages in the Nakdong river and its tributaries in South Korea between 2009 and 2018.

2.6.2 Research questions

To achieve the research aim, a series of research questions are addressed below.

- Research Question 1: How has water quality changed in the Nakdong river and its tributaries over the period 2001–2018?

  To address this research question, I will analyse water quality data using multivariate analyses to examine spatio-temporal changes in water quality in relation to climate and hydrological variables (Chapter 3). These analyses will help identify specific trends in water quality and any changes provoked by the construction of the barrages over the period from 2001 to 2018.

- Research Question 2: Are diatom samples reproducible at a reach scale to be used as a reliable means in water quality assessment?

  This question is asked to fill the gap in the use of a composite sample from multiple random samples at a reach scale. To examine variability in diatom assemblages at a reach scale, I will create the Replicability of Diatoms (ROD) dataset from fieldwork at three different sites in the Nakdong river and examine their similarity within the sites (Chapter 4). This analysis will demonstrate how similar diatom assemblages should be to be considered as replicate samples and what the implications are for water quality assessment.

- Research Question 3: Are diatom-based indices useful in demonstrating changes in water quality and the impact of the construction of the barrages over the period 2009–2018?

  To address this research question, I will review the results of the annual river health monitoring programme in the Nakdong river and examine the effectiveness of three common diversity indices with diatoms in demonstrating changes in water quality and the impact of the construction of the barrages on diatoms (Chapter 5). These analyses will provide strengths and weaknesses of diatom-based indices as a tools in river health assessment.

- Research Question 4: What are the relationships between water quality data and diatom data within the survey period?

  This research question is to address variability in diatom assemblages within the survey period. I will create the MY Diatoms (MYD) dataset by taking rock samples twice per site three weeks apart during autumn in the Nakdong river and its tributaries. Then, I will examine the variability between the two samples using multivariate analyses (Chapter 5). These analyses will demonstrate how important variability is within the period and what variable causes variability in diatom assemblages.

- Research Question 5: How have diatom assemblages responded to changes in the water quality and the construction of the barrages over the period 2009–2018?

  To address this question, I will use the Government Diatoms (GOVD) dataset that is collected in the Nakdong river and its tributaries and examine spatio-temporal changes in diatom assemblages from 2009 to 2018 using NMDS analysis and Community Trajectory Analysis (CTA) (Chapter 5). These analyses will visually demonstrate the responses of diatoms to environmental changes and enable those responses to be intuitively compared as geometric vectors.

- Research Question 6: What are species-environment relationships that can be used to better assess the health of rivers?

  To address this question, I will link water quality data and the GOVD data together in an attempt to holistically understand river health in the Nakdong river and its tributaries over the period 2009-2018 (Chapter 6). I will perform Cluster analysis, Indicator species analysis, and CCA analysis to achieve this goal. These analyses will provide a holistic understanding of river health based on relationships between water quality and diatoms. They should also provide an insight into how can river health be better assessed.

2.7 Thesis structure

This thesis consists of eight chapters in total including the Introduction in Chapter 1 and literature review, research aim, and questions in Chapter 2.

Chapter 3 addresses research question 1 by dealing with water quality data between 2001–2018 in the Nakdong river and its tributaries in relation to climate and hydrological variables, and the construction of the barrages. The examination of water quality in relation to those variables will not only facilitate an understanding of trends in water quality in time and space but also provide a basis for diatom analysis in Chapter 5.

Before examining the responses of diatoms over time in the Nakdong catchment in Chapter 5, Chapter 4 answers research question 2 by examining the replicability of diatom assemblages at a reach scale using the ROD dataset. I have found a gap in the use of a composite sample in water quality assessment, made from merging multiple random samples collected at a reach scale. However, variability in diatom assemblages at a reach scale has not been fully tested yet. Demonstrating the replicability of diatoms from random sampling at a reach scale will ultimately justify the current use of a composite sample as well as underpin the use of diatoms as a robust means for river health assessment.

Chapter 5 deals with diatoms in three parts. First, to answer research question 3, this chapter reviews the results of the mTDI and examines the effectiveness of three diversity indices with the GOVD. The examination of them will demonstrate their strengths and weaknesses in understanding river health in the Nakdong river and its tributaries. Second, this chapter analyses the MYD dataset to examine variability in diatom assemblages during autumn. This analysis will exhibit the extent of variability in the study area and identify what variables cause variability. Lastly, to tackle research question 5, this chapter constructs spatio-temporal changes of diatom assemblages from 2009 to 2018 using the GOVD dataset and compares their changes as geometric vectors in relation to water quality and the construction of the barrages.

In Chapter 6, water quality data and the GOVD are linked together to understand river health holistically in the Nakdong river and the tributaries from 2009 to 2018. An understanding of river health will be sought based on the groups defined by the statistical summary of water quality and indicator species. These analyses will be used to answer research question 6.

In Chapter 7, major findings from Chapter 3–6 are summarised and assessed to achieve the research aim. Then, the discussion moves on to how diatoms can be better used in river health assessment based on the findings and suggestions are made for detailed plans.

Finally, Chapter 8 discusses the implications of this research and closes with directions for future research to underpin the findings of this research project and extend this project in river health assessment.

Chapter 3

Water Quality Analysis

3.1 Introduction

This chapter looks at changes in water quality in the Nakdong river and its tributaries for last 18 years in relation to changes in the climate and hydrological system. These analyses are to answer research question 1, which is how has water quality changed in the Nakdong river and its tributaries over the period 2001–2018 (section 2.6.2).

There are four main tasks to be fulfilled in this chapter. First, the hydrological system in the study area will be examined to see spatio-temporal patterns of precipitation and discharge in the rivers. This will form the basis to understand the response of the flow regime to climate variables and the construction of the barrages in the main river. Second, spatio-temporal changes in water temperature in the rivers will be examined in comparison with trends in air temperature in the catchment. The results of water temperature will be discussed in relation to changes in air temperature as well as the construction of the barrages. Third, water quality parameters will be compared between the main river and the tributaries and mean change points in the trends of variables will be evaluated. Lastly, the relationships between water quality variables and sites will be established to understand the contribution of water quality variables and the distribution of variances in the Nakdong catchment. Ultimately, understanding spatio-temporal patterns of water quality in the study area in response to changes in climate and hydrological systems will provide the impact of the barrages on water quality and the fundamental basis to grasp the responses of diatoms in the rivers.

3.2 Methods

To achieve the tasks above, methods, datasets, and expected outcomes are presented in the flow chart (Figure 3.1). More detailed methods are explained below.

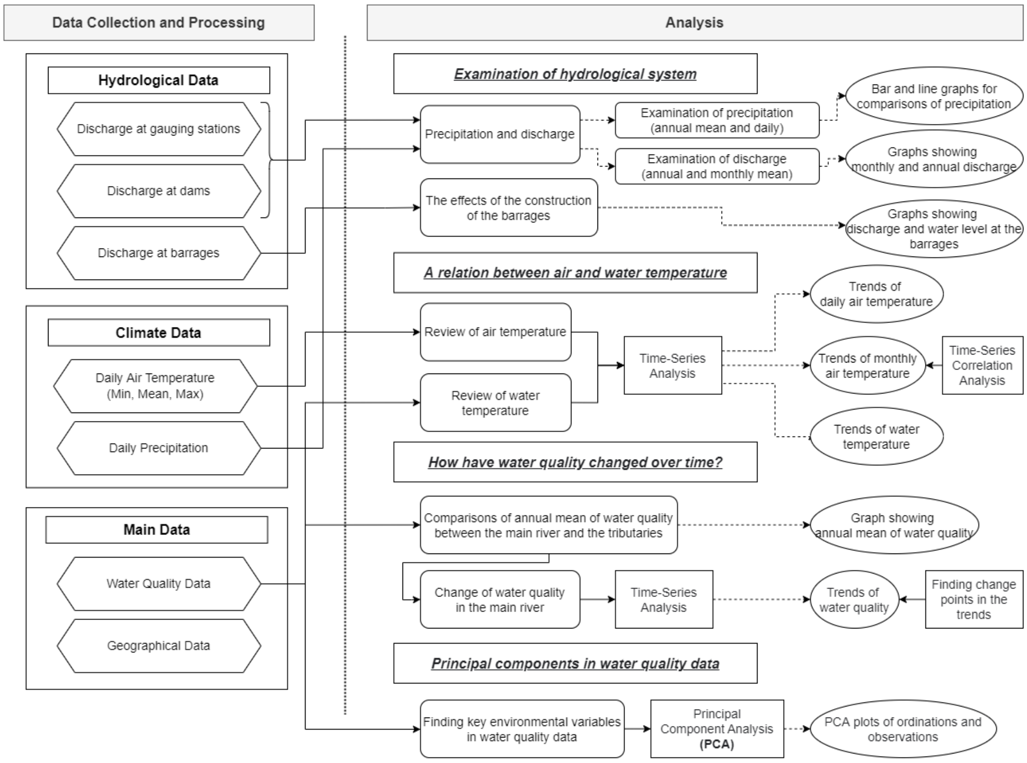


Figure 3.1: Data requirements and workflow for water quality analysis

3.2.1 Examination of hydrological system

In order to examine the hydrological system in the main river and the tributaries from 2001 to 2018, the relationship between precipitation and discharge was investigated. First, annual precipitation plots and cumulative precipitation plots at 16 weather stations in the Nakdong catchment were created to evaluate meteorologically dry and wet seasons. The impact of dry and wet seasons on discharge was then reviewed with hydrographs by catchments. Next, annual and monthly mean discharge plots in the main river and the tributaries were generated to see the responses of discharge to the meteorological dry conditions and the construction of the barrages. In addition, plots of discharge and water level at the barrages in the main river were produced to pinpoint the timing of the operation and their impacts on the hydrological system.

3.2.2 A relation between air and water temperature

Water temperature is important because it influences on water chemistry and biological communities in rivers. Understanding of water temperature should be sought in consideration of air temperature because the water temperature is closely related to air temperature (Boyd, 2019). Therefore, changes in water temperature from 2001 to 2018 in the main river and the tributaries were examined with trends in air temperatures in the Nakdong catchment.

First of all, time-series trends of daily maximum, mean, and minimum air temperatures in the catchment were constructed to see overall trends after the data was broken down using the decompose command in the Stats package (R Core Team, 2020). Similarly, monthly time-series trends of air temperatures in the catchment were compared in matrices and dendrograms to see spatial correlations between weather stations. For this analysis, monthly air temperatures were calculated based on daily air temperatures per weather station, and a correlation-based dissimilarity function (the diss command with correlation method) in the TSClust package (Montero and Vilar, 2014) was used.

Afterwards, time-series and seasonal trends of monthly water temperatures in the rivers were obtained by site using the same process. The obtained trends of monthly water temperatures were tested using the Mann-Kendall Trend Test (the mktest command) in the Trend package (Pohlert, 2020) to see if they have any monotonic trends. If so, a change point was sought using the Pettitt's test (Pohlert, 2020). Seasonal trends of monthly water temperatures were also tested using the seasonal Mann-Kendall Trend Test using the smktest command to evaluate changes by comparing the same month each year. Then, the results of monthly water temperature trends in the rivers were discussed in relation to the results of air temperature trends and the construction of the barrages. In the process of monthly averaging of water temperatures, there were 14 and 37 NAs in the main river and the tributaries, which were all linearly interpolated using the na_interpolation command in the ImputeTS package (Moritz and Bartz-Beielstein, 2017).

3.2.3 Changes in water quality

Water quality in the Nakdong river and its tributaries was examined to understand spatio-temporal changes in water quality in the catchment from 2001 to 2018. First of all, in order to understand annual trends of water quality in the main river and the tributaries, the annual means of nine different types of water quality variables were illustrated. Then, water quality in the main river was further scrutinised in time and space to see how they have changed regarding the construction of the barrages. Water quality variables in the main river were plotted in the time-series order from 2001 to 2018 and multiple change points in mean and variance were sought in the trends of the variables by the site. For this analysis, the cptmeanvar command with binary segmentation method in the Changepoint package (Killick et al., 2016) was used. In the process of obtaining the monthly mean values, NAs were linearly interpolated.

3.2.4 Finding the principal component in water quality

In an attempt to find the dominant variables in water quality data, Principal Component Analysis (PCA) was performed for all sites in the main river and the tributaries using the prcomp command in the Stats package (R Core Team, 2020). Data used for this analysis were all observations of water quality variables recorded from 2001 to 2018 in the rivers and any observations containing NAs (943 out of 8746 in the main river and 722 out of 7802 in the tributaries) were removed to minimise errors.

3.3 Datasets

In this chapter, water quality data, hydrological data, and climate data from relevant administrations in South Korea were used. More details about the data were explained below and their locations are presented in Figure 3.2.

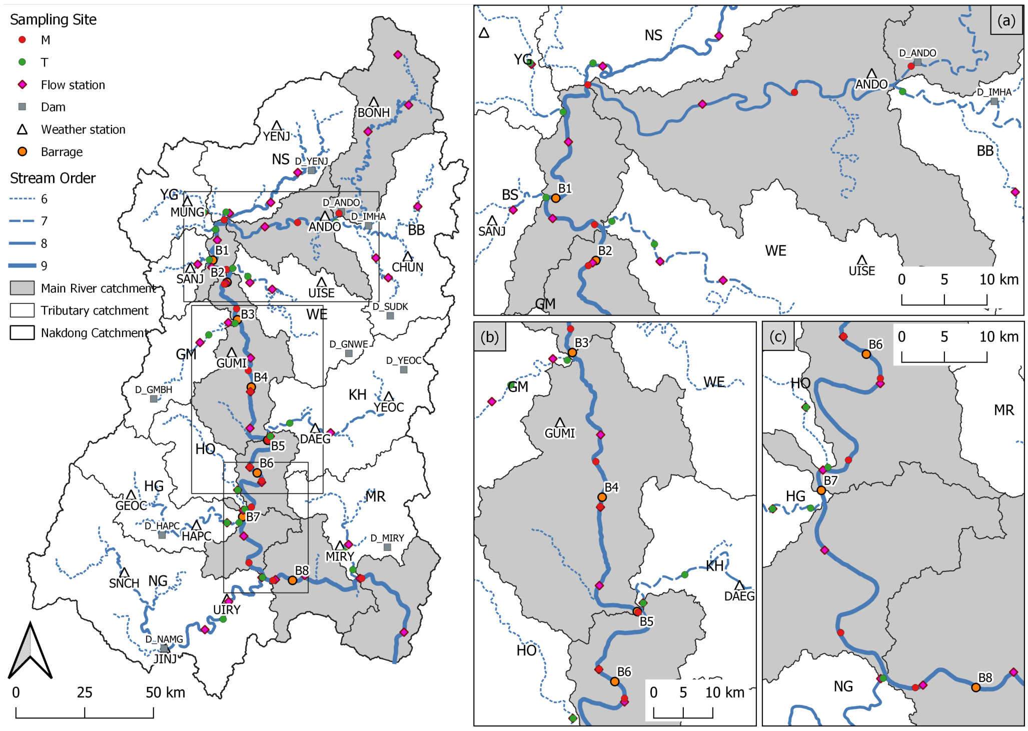


Figure 3.2: Map showing locations of water quality sampling sites, flow stations, weather stations, dams, and barrages in the study area. (a), (b), and (c) are inset maps at a large scale for parts of the study area. Two-letter words represent the name of the river and the catchment in the tributaries.

3.3.1 Water quality data

Water quality data used are secondary data maintained by the National Institute of Environmental Research (NIER) in South Korea across 34 sites (Sampling site M and T in Figure 3.2 and Table 3.1). Data coverage and resolution vary from site to site within a range of 2001 to 2018, but all observations have been measured weekly since the mid-2000s, before then they were monthly collected. Parameters included are water temperature (WT, °C), pH, dissolved oxygen (DO, mg/L), biological oxygen demands (BOD, mg/L), chemical oxygen demand (COD, mg/L), total nitrogen (TN, mg/L), total phosphorus (TP, mg/L), suspended solids (SS, mg/L), electrical conductivity (EC, $\mu S\,cm^{-1}$), and chlorophyll-a (Chla, mg/L). To raw water temperature values, 3 was added to make them positive for convenience of calculation. Except for water temperature and pH, other variables were transformed via log or square-root or cube-root to make them follow the normal distribution. All data were downloaded from the Water Environment Information System (2019) and a statistical summary of the raw data is presented in Table 3.2.

Table 3.1: Water quality sampling sites and barrages. M and T in River type represent the main river and the tributaries. Stream order data were obtained from the Water Environment Information System (2019) and the distance was calculated in QGIS.

Site	River type	River name	Strahler Stream order	Distance (km) from the river mouth
M1	M	ND	7	343.02
T1	T	BB	7	341.22
M2	M	ND	8	320.15
M3	M	ND	8	280.35
T2	T	NS	8	283.86
T3a	T	YG	6	284.76
T3b	T	YG	7	274.00
B1	Barrage	ND	9	259.77
T4	T	BS	7	258.72
M4	M	ND	9	248.87
T5a	T	WE	7	258.63
T5b	T	WE	7	249.47
B2	Barrage	ND	9	243.22
M5	M	ND	9	242.06
M6	M	ND	9	228.94
B3	Barrage	ND	9	224.34
T6a	T	GM	6	239.42
T6b	T	GM	6	224.64
M7	M	ND	9	201.79
B4	Barrage	ND	9	195.14
M8	M	ND	9	193.50
B5	Barrage	ND	9	169.00
M9	M	ND	9	168.86
T7a	T	KH	7	183.65
T7b	T	KH	7	170.60
M10	M	ND	9	152.63
B6	Barrage	ND	9	149.02
M11	M	ND	9	145.47
M12	M	ND	9	125.52
T8a	T	HO	6	130.69
T8b	T	HO	6	122.27
B7	Barrage	ND	9	119.05
T9a	T	HG	7	125.62
T9b	T	HG	7	119.39
M13	M	ND	9	95.87
T10a	T	NG	8	123.18
T10b	T	NG	8	87.97
M14	M	ND	9	83.68
B8	Barrage	ND	9	75.24
T11a	T	MR	7	59.68
T11b	T	MR	7	50.72
M15	M	ND	9	44.43

Table 3.2: Summary statistics of water quality data from 2001 to 2018 in the Nakdong catchment. Main river represents the Nakdong river and BB–MR are the name of the rivers for the tributaries. See the locations in Figure 3.2.

Variable		Total (N = 16,548)	Main River (N = 8,746)	All Tributaries (N = 7,802)	BB (N = 581)	NS (N = 581)	YG (N = 432)	BS (N = 446)	WE (N = 661)	GM (N = 751)	KH (N = 1,279)	HO (N = 661)	HG (N = 663)	NG (N = 1,011)	MR (N = 736)
	Min	-2	-2	-1	-1.0	-1.0	0.2	0.0	0.0	0.0	-1.0	0.0	0.1	0.0	1.0
WT	Max	36	36.0	35.4	31.6	33.1	31.2	34.0	35.4	33.4	33.8	35.4	33.1	34.0	34.1
(℃)	N;Mean±sd	16,545; 16.32±8.44	8,745; 16.02±8.43	7,800; 16.65±8.44	15.17±8.38	15.45±9.38	15.91±8.76	16.91±8.29	16.72±9.10	17.46±8.19	17.28±8.19	17.61±8.55	662; 15.54±7.40	16.90±8.43	735; 16.83±8.12
	Min	5.4	5.4	5.5	5.9	6.2	6.1	6.6	6.4	6.4	6.4	6.4	6.1	5.9	5.5
pH	Max	10.8	10.8	9.7	9.1	9.6	9.7	9.4	9.5	9.4	9.6	9.7	8.8	9.5	9.6
	N;Mean±sd	16,545; 7.93±0.55	8,745; 7.98±0.58	7,800; 7.88±0.52	7.76±0.38	7.78±0.43	7.80±0.58	7.80±0.46	7.95±0.44	7.77±0.38	8.05±0.55	8.03±0.52	662; 7.75±0.35	7.90±0.55	735; 7.81±0.69
	Min	2.6	3.1	2.6	6.3	4.8	4.5	6.0	3.2	6.6	3.9	5.4	4.5	6.0	2.6
DO	Max	24.9	24.9	19.6	16.2	18.2	17.9	17.9	18.5	16.9	18.1	18.3	16.7	18.2	19.6
(mg/L)	N;Mean±sd	16,542; 10.75±2.52	8,742; 10.83±2.68	7,800; 10.67±2.32	10.63±2.26	10.72±2.43	11.04±2.47	10.73±2.13	10.84±2.43	9.94±1.88	10.93±2.59	11.06±2.38	662; 10.37±1.92	10.53±2.16	735; 10.71±2.40
	Min	0.1	0.3	0.1	0.3	0.2	0.2	0.3	0.3	0.3	0.7	0.2	0.1	0.4	0.2
BOD	Max	12.5	8.4	12.5	7.0	8.5	7.6	12.5	7.9	8.4	12.4	6.1	4.3	7.7	7.9
(mg/L)	N;Mean±sd	16,545; 1.89±1.25	8,745; 1.91±1.02	7,800; 1.87±1.46	1.15±0.63	0.86±0.80	1.07±0.71	1.65±1.21	1.73±0.95	1.38±0.99	3.64±1.72	1.36±0.75	662; 0.80±0.48	2.48±1.25	735; 2.00±1.28
	Min	0.4	1.2	0.4	2.8	1.3	0.4	2.2	1.8	2.0	3.6	1.6	0.6	2.4	0.7
COD	Max	38.3	22.5	38.3	37.0	38.3	7.0	17.0	17.0	25.4	17.8	19.7	12.1	16.8	14.2
(mg/L)	N;Mean±sd	16,545; 5.36±2.19	8,745; 5.51±1.80	7,800; 5.18±2.54	5.36±2.01	3.37±2.64	2.65±1.03	5.12±2.05	5.56±2.19	4.67±1.87	8.40±2.12	4.10±1.57	662; 3.31±1.09	5.85±1.81	735; 4.33±1.73
	Min	0.313	0.600	0.313	0.643	0.862	0.364	1.048	0.451	1.792	2.804	0.313	0.469	0.665	0.427
TN	Max	18.384	8.518	18.384	4.836	5.719	4.838	7.579	6.149	7.819	18.384	4.992	6.038	9.790	5.879
(mg/L)	N;Mean±sd	15,940; 3.05±1.59	8,142; 2.80±0.99	7,798; 3.30±2.01	2.06±0.65	3.50±0.83	2.08±0.70	3.08±1.27	2.62±1.09	750; 4.02±1.04	1,278; 6.67±1.91	2.00±0.81	662; 1.68±0.42	2.58±1.42	735; 2.57±0.84
	Min	0	0	0	0.004	0.008	0.000	0.018	0.004	0.011	0.027	0.004	0.000	0.007	0.000
TP	Max	2.049	0.976	2.049	0.427	0.684	0.544	0.745	0.245	0.675	2.049	0.385	0.335	0.557	0.653
(mg/L)	N;Mean±sd	15,940; 0.09±0.13	8,142; 0.08±0.07	7,798; 0.11±0.16	0.02±0.02	0.07±0.07	0.03±0.04	0.11±0.10	0.04±0.04	750; 0.11±0.10	1,278; 0.36±0.24	0.04±0.03	662; 0.04±0.03	0.08±0.06	735; 0.07±0.06
	Min	0.1	0.2	0.1	0.3	0.2	0.1	0.4	0.2	0.4	0.8	0.3	0.3	0.4	0.4
SS	Max	560	560	498	414.0	498.0	32.8	356.0	179.1	217.0	221.0	185.0	172.0	178.0	191.0
(mg/L)	N;Mean±sd	16,545; 12.86±21.44	8,745; 14.02±22.19	7,800; 11.55±20.49	6.47±19.70	16.85±44.06	2.90±3.60	14.37±31.11	10.59±17.27	13.69±20.52	11.76±13.42	7.69±13.48	662; 15.15±15.61	14.52±14.76	735; 9.22±13.88
	Min	3	3	19	80	70	37	103	71	90	108	19	59	22	60
EC	Max	7789	7789	4200	1040	455	733	1382	600	611	1410	605	268	628	4200
($\mu S\,cm^{-1}$)	N;Mean±sd	16,545; 287.16±178.14	8,745; 276.92±150.03	7,800; 298.64±204.53	189.41±44.87	252.54±55.11	267.26±84.45	314.60±153.79	290.13±69.15	317.27±91.58	628.65±248.26	214.97±76.70	662; 128.34±31.18	208.06±94.58	735; 197.77±165.30
	Min	0	0	0	0.6	0.2	0.0	0.7	0.3	0.1	2.2	0.5	0.0	1.9	0.7
Chl-a	Max	258.3	251.3	258.3	67.0	74.8	41.3	154.8	89.9	126.8	258.3	62.6	25.2	176.4	187.2
(mg/L)	N;Mean±sd	15,183; 19.71±25.80	8,153; 22.70±26.14	7,030; 16.24±24.96	5.45±5.75	5.21±7.91	382; 3.24±4.48	401; 8.38±11.88	592; 10.70±10.46	569; 7.12±10.22	1,275; 37.94±36.62	591; 7.75±7.30	591; 3.54±2.72	830; 31.79±31.26	637; 18.14±23.31

3.3.2 Hydrological data

The hydrological dataset includes discharge data at 34 gauging stations, discharge data at ten dams, and discharge and water levels at eight barrages in the catchment (see the locations in Figure 3.2). The numbers of gauging stations are 14 and 20 for the main river and the tributaries and the data roughly ranges from 2006 to 2018. Because of the discontinuity of observations at some gauging stations, multiple stations in each tributary were sometimes included to supplement each other if necessary. In the process of monthly averaging, NAs were removed. Data at gauging stations were downloaded at the Han River Flood Control Office (Han River FCO, 2020), while other data for dams and barrages were downloaded from the Korea Water Resources Corporation (K-Water, 2020).

3.3.3 Climate data

The climate data are composed of daily precipitation (mm) and daily maximum, mean, and minimum temperatures (℃) recorded at 16 weather stations from 2001 to 2018 in the Nakdong catchment (see the locations in Figure 3.2). SANJ station has a data coverage from 1st Jan 2002, while observations at CHUN and UIRY stations started from 4th Sep 2010 and 22nd June 2010, respectively. Because of different lengths of data, CHUN and UIRY stations were removed in the Time-series clustering analysis. All data were downloaded from the Korea Meteorological Administration Web portal (KMA, 2020).

3.4 Results

3.4.1 Examination of the hydrological system

3.4.1.1 Precipitation and discharge

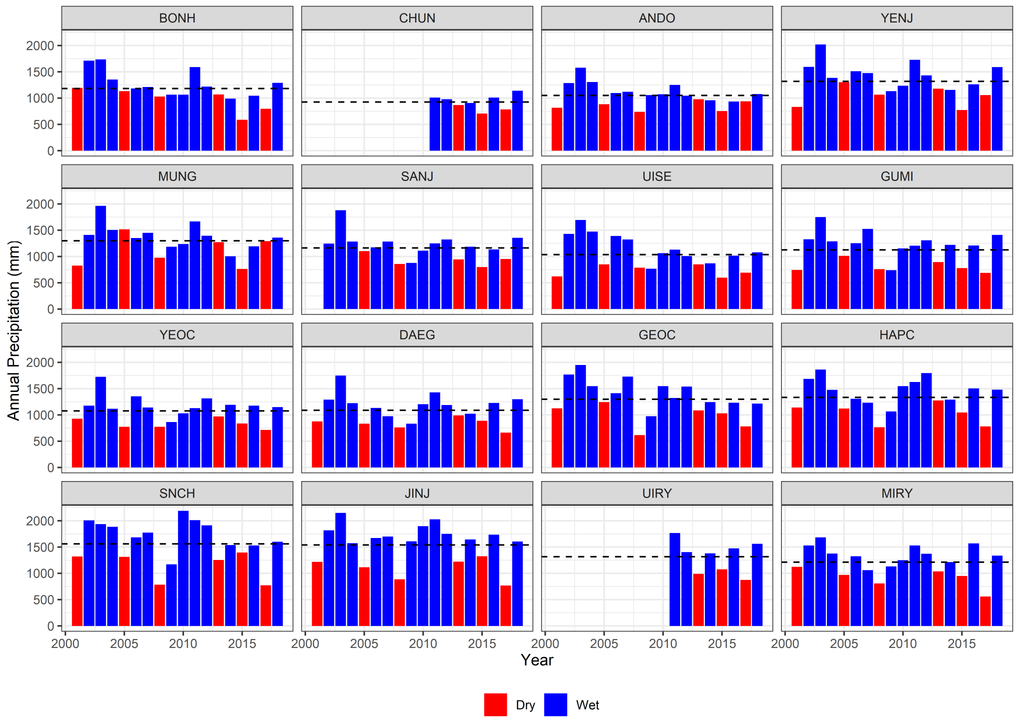


Figure 3.3: Annual and mean precipitation at 16 weather stations from 2001 to 2018. Red bars and blue bars indicate dry and wet years each, while dashed lines present annual mean precipitation during the period at each station. The order of the stations is roughly laid out from the north to the south. For more detailed locations, see Figure 3.2.

Figure 3.3 shows annual precipitation and average annual precipitation at each station in the study area from 2001 to 2018. Mean annual precipitation broadly ranges from 1000 mm to 1500 mm with spatial variations. Stations in the south like SNCH and JINJ tend to have higher precipitation at around 1500 mm, while BONH, ANDO, and YENJ in the north have around 1200 mm. In between, midland regions such as YEOC and DAEG have the least precipitation which is just above 1000 mm. These spatial patterns of precipitation are typically constrained by the

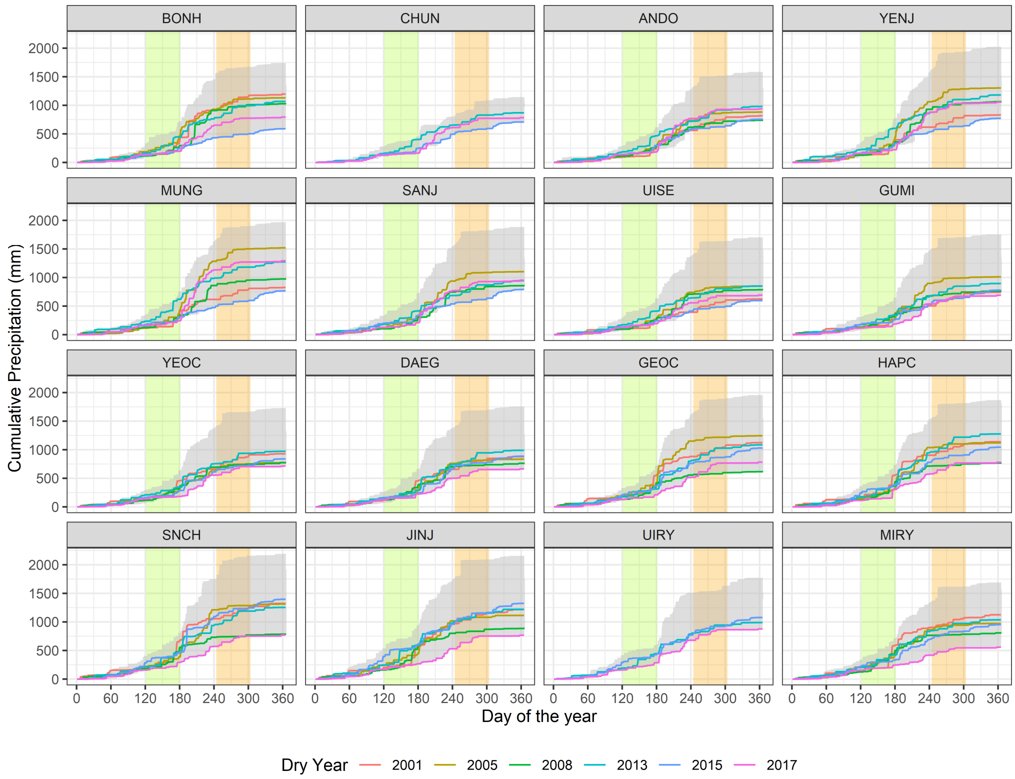


Figure 3.4: Cumulative annual precipitation at 16 weather stations. Six dry years (2001, 2005, 2008, 2013, 2015 and 2017) are plotted in different colours. The grey shade for each plot represents the distribution of cumulative annual precipitation from 2001 to 2018. The green and amber coloured vertical rectangles represent the diatom survey periods in the national monitoring programme for spring (May–June) and autumn (Sep–Oct).

effect of orography and distance from the coastline.

Over the last 18 years, meteorological dry conditions were much more dominant in the 2010s than in the 2000s and there have been roughly six dry calendar years observed: 2001, 2005, 2008, 2013, 2015, and 2017 (see the red bars in Figure 3.3). Among them, droughts in 2008, 2015, and 2017 were the most severe events. The 2008 event was extensively documented across the study area and prolonged to 2009 in some stations, while the recent two drought events differently affected the north and the south in the study area. For example, BONH, CHUN, ANDO, YENJ, MUNG, and SANJ in the north recorded the driest year in 2015, while SNCH, JINJ, UIRY, and MIRY in the south did in 2017. In between, UISE, GUMI, YEOC, and DAEG showed a fairly similar amount of precipitation received in 2015 and 2017.

One of the features of precipitation in South Korea is a high concentration of rainfall in summer due to the monsoon and typhoons and this feature is well illustrated by the exponential growth of slope between spring and autumn (see the grey area between two diatom survey periods in Figure 3.4). However, during the recent two dry years 2015 and 2017, there was little addition of precipitation in the summer, resulting in a low and gentle slope of graphs (see the lines for 2015 and 2017 in Figure 3.4). Figure 3.4 also exhibits that lack of precipitation in 2015 was much more significant in the north of the study area than in the south, contrarily, the year 2017 had the exact opposite pattern.

Depending on summer precipitation, the impact of meteorological drought would be considerably different within the year. In spring, there is little difference between dry years and non-dry years, while the gaps between them can be much wider in autumn. As a result, autumn would have more severe drought conditions than spring unless there was sufficient precipitation in summer supplies. In case the dry period continues over the winter, drought conditions in spring could be even worse before the monsoon season comes in summer. For example, some stations like UISE, GUMI, DAEG, GEOC, HAPC, and SNCH had prolonged dry periods over 2008 winter during the two dry years 2008–2009.

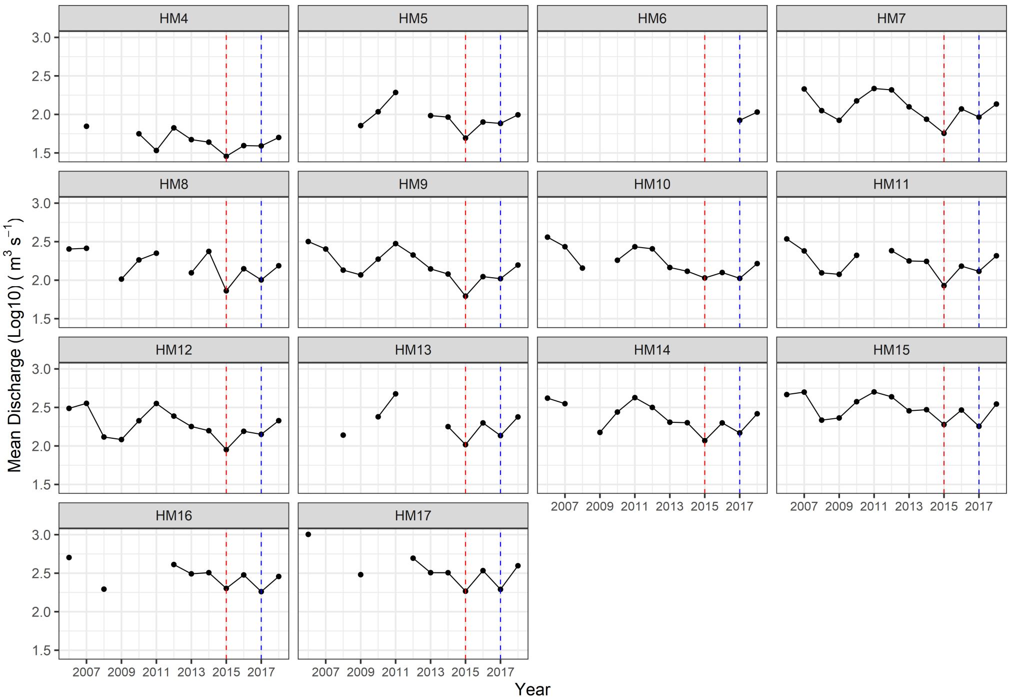


Figure 3.5: Annual mean discharge in the main river from 2006 to 2018. The red dashed lines indicate the dry year of 2015 and the blue dashed lines indicate the dry year of 2017. See the location of flow gauging stations in Figure 3.2.

The spatio-temporal patterns of meteorological droughts are well-matched with hydrological droughts in annual mean discharge in the rivers (Figure 3.5 and 3.6). Annual mean discharges in the main river and the tributaries tend to be lower in the 2010s than in the 2000s. Also, hydrological droughts in 2015 and 2017 are well-pitched in different spatial extents divided by north and south regions with the drought in 2008 in common. The degree of hydrological droughts would be more significant in the tributaries than in the main river. For example, although the main river had three low discharges that all correspond to the meteorological droughts, nevertheless, there was sufficient discharge flowing in the main river during that time. The tributaries recorded similar features, but small rivers and streams would suffer relatively more than big rivers. For instance, tributaries BB and BS would be running almost dry during the droughts, while relatively big rivers like KH, HG, NG, and MR were running low but not as entirely dry as the small tributaries.

The distributions of monthly discharge by year in the main river are presented in

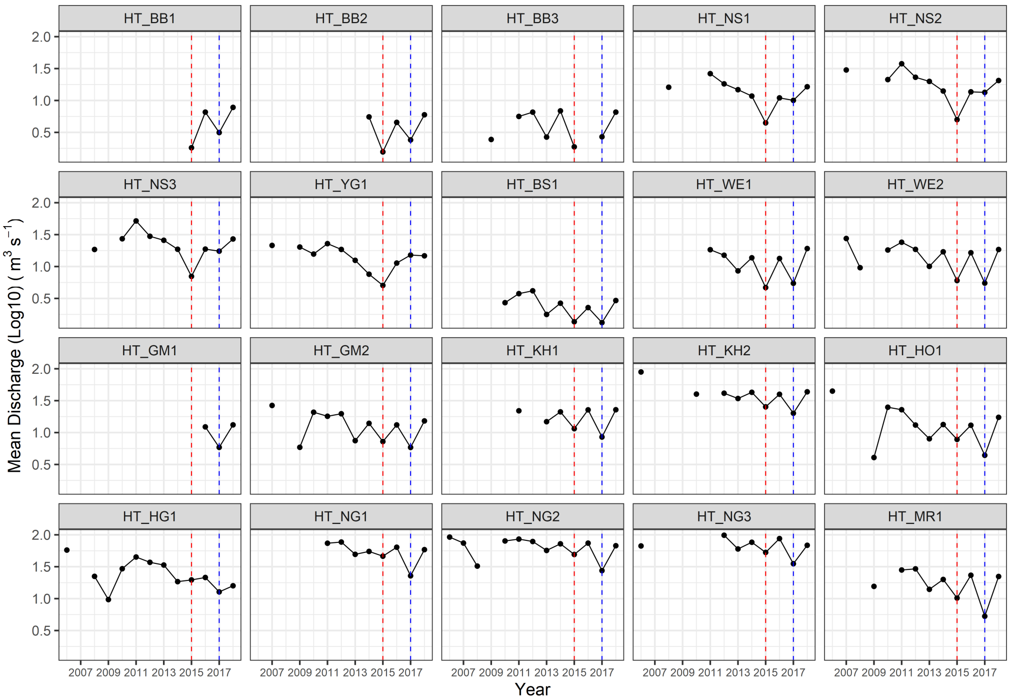


Figure 3.6: Annual mean discharge in the tributaries from 2006 to 2018. Red dashed lines indicate the dry year of 2015 and blue dashed lines indicate the dry year of 2017. See the location of flow gauging stations in Figure 3.2.

Figure 3.7 and it shows some notable changes over time. Owing to the summer concentration of precipitation, the distribution of monthly discharge generally forms a peak in summer, particularly in July and August. Such a peak can emerge in September or early October because of the influence of typhoons. For example, the peaks of September in 2007, 2010, and 2012 were almost as high as those of July or August in the same year or higher than those.

Since 2012, however, the distribution of monthly discharge in the main river has changed with the suppression of the summer peak and the escalated discharge in the winter. The lowered summer peaks in 2015 and 2017 may be explained by the smaller addition of summer precipitation in those years, nevertheless, the summer peaks in other years have been notably reduced. In contrast, during the winter, monthly discharge in the main river has maintained at a stable and higher level without fluctuating much. For example, the winters in 2011 to 2012, to 2013, had as much discharge as spring or autumn in those years.

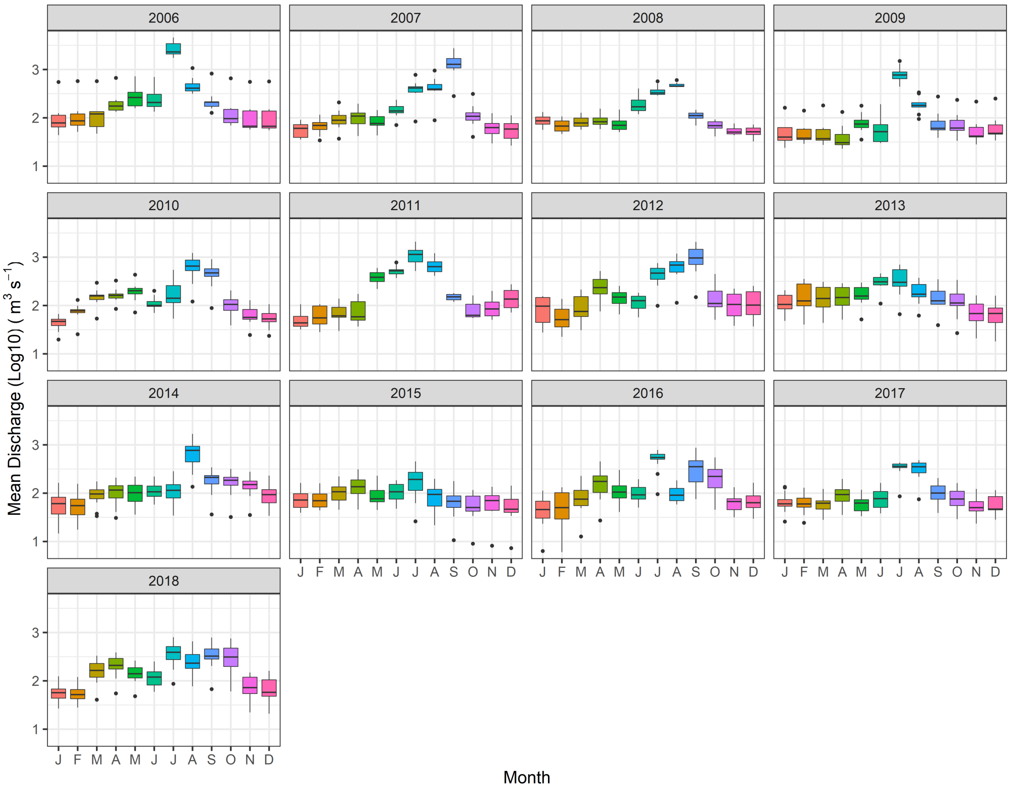


Figure 3.7: Monthly discharge in the main river from 2006 to 2018.

The distribution of monthly discharge for the tributaries varies by rivers and its catchment (see Figure B.1–B.9 in Appendix B). There were no such systematic changes as the reduction of the summer peak or the increased winter discharge observed in the main river, but the distribution of monthly discharge has remained similar unless there was a meteorological drought. For example, there was a clear disappearance of the summer peak due to meteorological droughts in 2015 and 2017.

3.4.1.2 The effects of the barrage construction

To understand the effects of the barrage constructions on the hydrological system, the timing of the barrage construction and operation and its impacts on the flow regime in the main river were investigated using water level data at reservoirs created above the barrages and discharge data at the barrages.

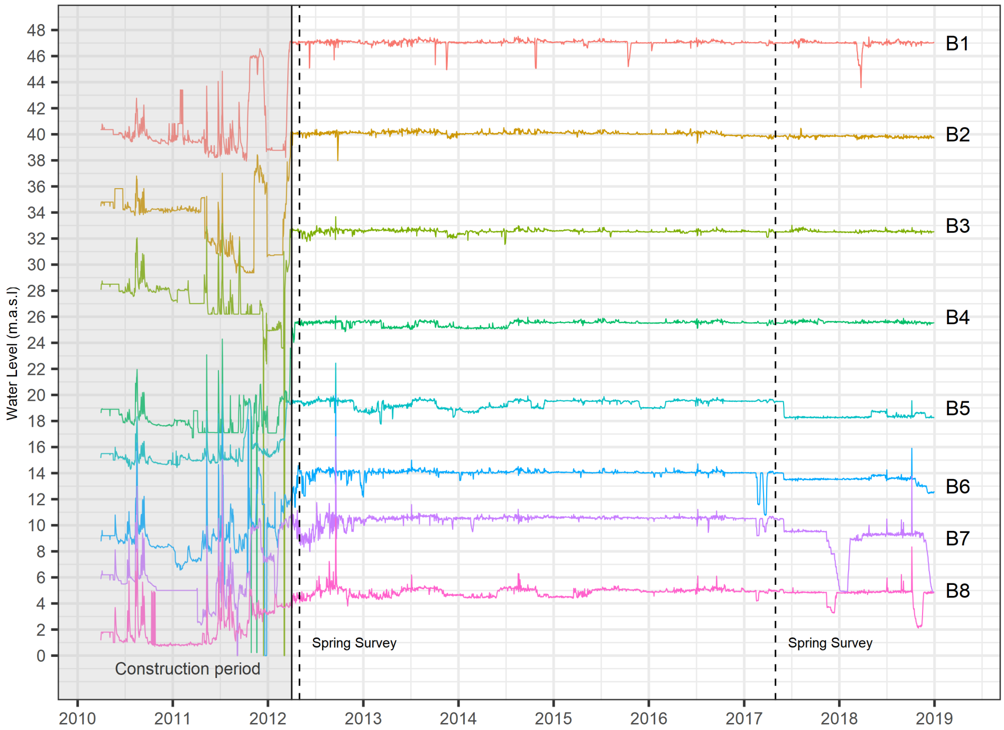


Figure 3.8: Water level at the barrages in the main river from 2010 to 2018. The grey area indicates the construction period and the black solid line means the end of the construction in April 2012. The black dashed lines indicate the start of the spring diatom survey period (May–June) in 2012 and 2017, respectively.

Figure 3.8 shows the change in water level at the location of all eight barrages in the main river from 2010 to 2018. Although there were natural rising and falling limbs in the water level during the summer in the first two years of the construction period, the change in water level was considerably disrupted by the construction. For instance, from mid-2010, the water level sharply increased or dropped or levelled off unnaturally until the start of 2012. From March 2012, the water level at the barrages started to creep up and had reached the planned altitude by around April before the spring diatom survey was carried out in May 2012. Afterwards, despite the water level moving up and down at the barrages, they remained stable without significant change until the summer of 2017. In June 2017, the water level at three barrages (B5, B6, and B7) has been lowered by 1.5, 0.5 and 1.0 m, respectively following the authority's decision to monitor the effect of gates being left open on water quality and the riverine ecosystems.

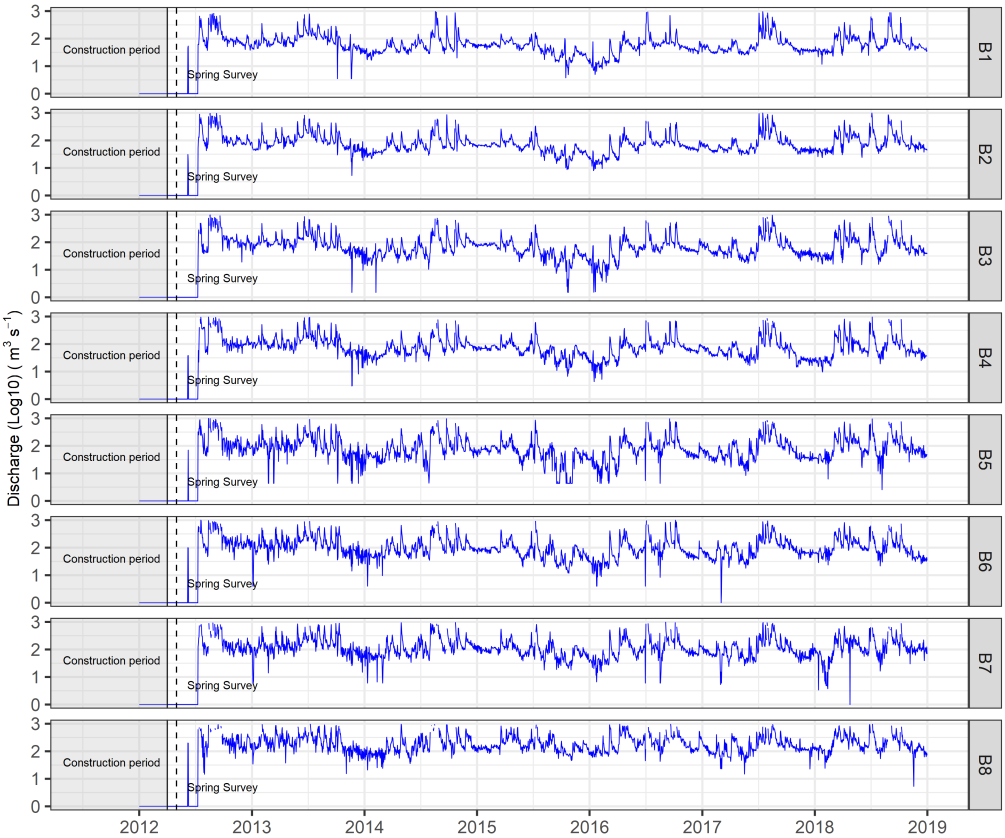


Figure 3.9: Discharge at the barrages in the main river from 2012 to 2018. The grey area indicates the construction period and the black solid lines mean the end of the construction in April 2012. The black dashed lines indicate the start of the spring diatom survey period (May–June) in 2012.

Discharge at the barrages shows how each barrage has behaved in controlling flow in the main river (Figure 3.9). Once they were completed in April 2012, the barrages continued to store water without flowing down until July 2012 when the monsoon season started. Afterwards, all eight barrages have been constantly regulating the flow of the main river similarly.

3.4.2 Trends of air temperature and water temperature

To examine changes in water temperature in the main river and the tributaries, air temperatures in the Nakdong catchment were examined to construct overall time-series trends of air temperatures. Then, monthly air temperatures were examined to see spatio-temporal patterns of air temperatures in the catchment. Afterwards, time-series trends of monthly water temperatures in the rivers were produced to understand changes in water temperatures from 2001 to 2018. The obtained trends were then tested to see if they have monotonic trends with a change point. In addition, seasonal trends of water temperature by the same month each year were also evaluated to see seasonal changes in water temperature in space.

3.4.2.1 Air temperature

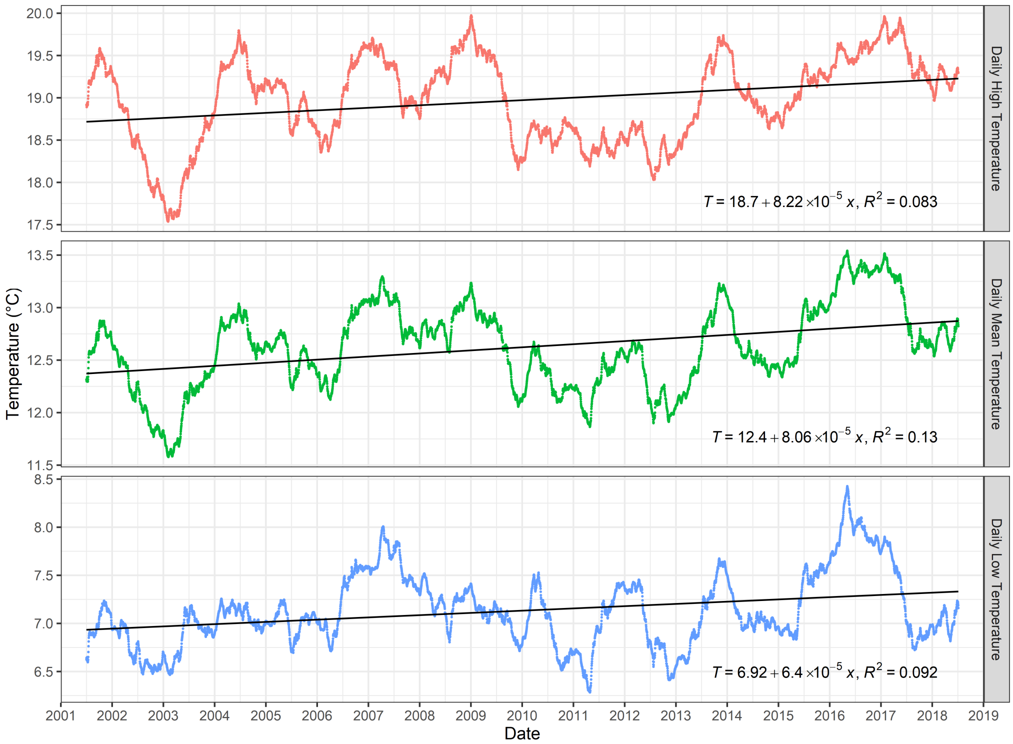


Figure 3.10: Time-series trends of three types of daily air temperatures in the Nakdong catchment; maximum, mean, and minimum temperatures. Slope for each linear regression equation is per day.

Time-series trends of daily maximum, mean, and minimum temperatures from 2001 to 2018 are presented in Figure 3.10 after being extracted from averaged time-series daily temperatures from 16 weather stations in the Nakdong catchment. Throughout this time, the three daily temperatures have fluctuated up and down similarly. In the early 2000s, three daily air temperatures were low in the period from 2002 to 2003 and they remained high from 2006 to 2009. Again, they dropped and remained low until 2013. Afterwards, they started to rise and maintained a high during 2013–2017 with the highest peak recorded in the summer of 2016. Later, they slightly came down in the summer of 2017.

The observed time-series trends in three daily air temperatures were tested to see if there is a monotonic upward or downward trend. The linear regression results show that daily maximum, mean, and minimum temperatures have been on an upward trend ($p < 0.05$) with an increase of 0.03, 0.029, and 0.023 ℃ per year in the study area between 2001 and 2018 (see black solid lines in Figure 3.10).

Next, to understand spatio-temporal patterns of air temperatures in the catchment, a time-series clustering analysis was performed using monthly maximum, mean, and minimum temperatures at 14 weather stations. A dissimilarity matrix was calculated based on the correlation method and it ranges from 0 to 2. The results show that the difference in the time-series trends of air temperatures between the weather stations is marginal at a range from 0 to below 0.1 in all three types of monthly air temperatures (Figure 3.11). The correlation between the weather stations is widely constrained by geographic location (latitude) with a north-south divide (Figure 3.12). For example, the monthly maximum temperature consists of two groups that are geographically divided by the stations in the north and in the south (red and green in Figure 3.12 (a)). The monthly mean temperature has roughly three groups with a north-south divide and one distinguishable weather station MIRY (blue in the north and green in the south in Figure 3.12 (b)). In contrast, the monthly minimum temperature has little geographical distinction with two noticeable weather stations DAEG and MIRY separated from others (Figure 3.12 (c)). These north-south patterns in monthly maximum, mean, and minimum air temperatures suggest that the distribution of air temperatures in the Nakdong catchment primarily geographically depends on latitude and the difference in the monthly minimum air temperature is relatively smaller than those in the maximum and mean air temperatures.

To sum up, daily maximum, mean, and minimum air temperatures in the Nakdong

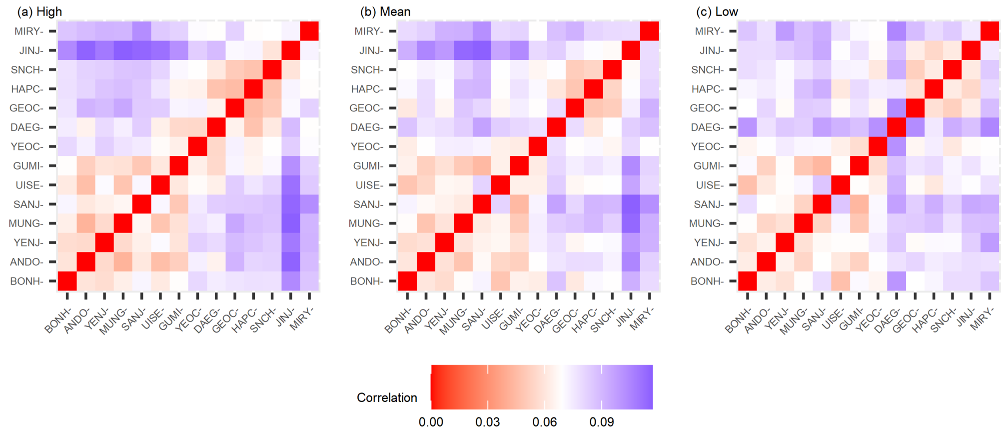


Figure 3.11: Dissimilarity matrices for time-series trends of monthly air temperatures in the study area. (a) Monthly maximum temperature, (b) Monthly mean temperature, (c) Monthly minimum temperature. The order of weather stations on both axes is laid out based on the geographical location from the north to the south. In this analysis, two weather stations CHUN and UIRY were excluded due to the different lengths of data coverage.

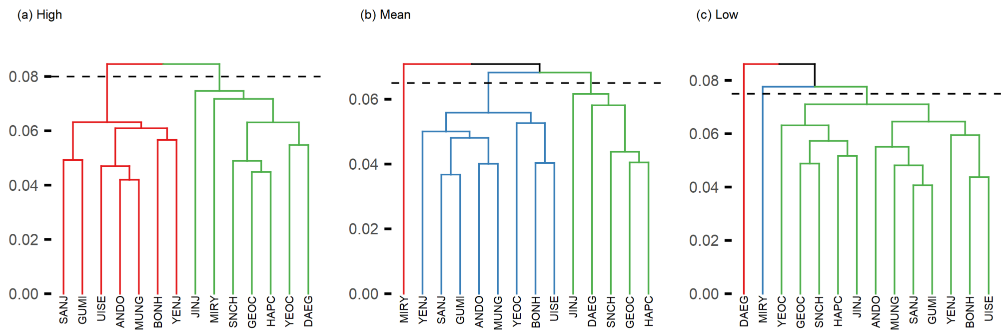


Figure 3.12: Dendrograms for time-series trends of monthly air temperatures in the study area. (a) Monthly maximum temperature; weather stations in red are from the north and those in green are from the south, (b) Monthly mean temperature; weather stations in blue are from the north and those in green are from the south with MIRY in red from the south, (c) Monthly minimum temperature; Two weather stations DAEG and MIRY are separate from others. In this analysis, two stations CHUN and UIRY were excluded due to the different lengths of data coverage.

catchment have been on an upward trend with ups and downs over the last 18 years; temperatures were high during the late 2000s before dropping to remain low in the early 2010s. From 2013, temperatures rose to reach a peak in 2017 before dropping in summer 2017. The results of time-series clustering analysis for monthly air temperatures show that air temperatures in the catchment are spatially correlated with a clear north-south divide, constrained by latitude.

3.4.2.2 Water temperature

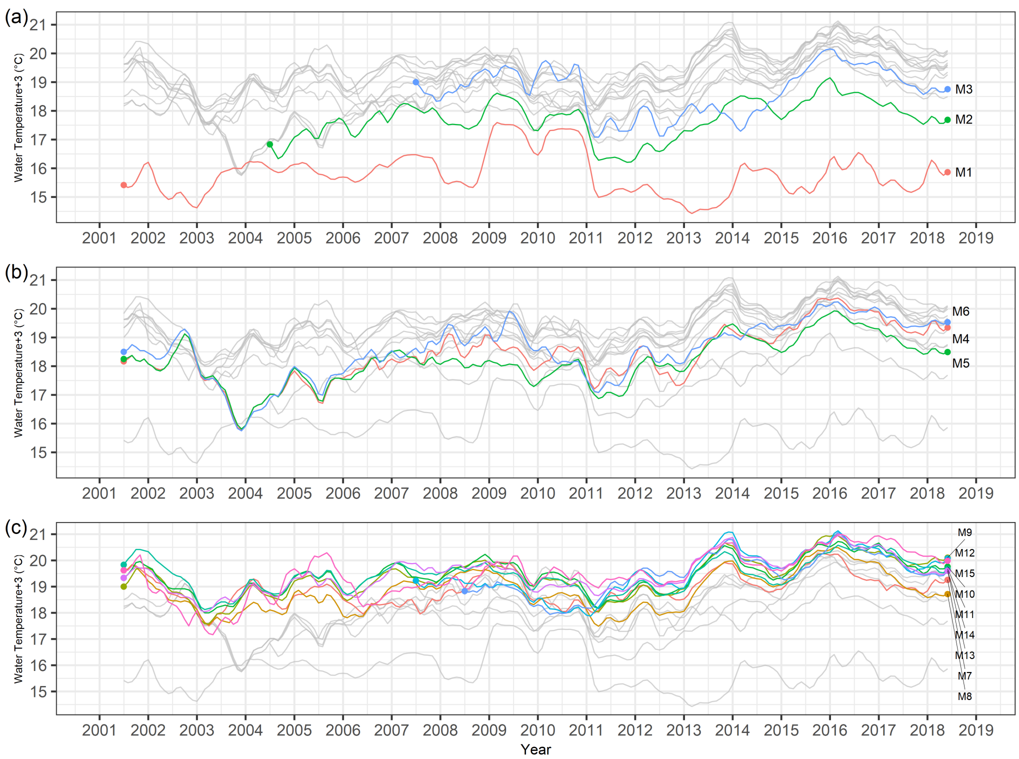


Figure 3.13: Time-series trend of monthly water temperature in the main river. (a) trends in sites M1–M3, (b) trends in sites M4–M6, (c) trends in M7–M15. Note: the starting point for each site differs.

Time-series trends of monthly water temperature in the main river obtained from decomposed time-series analysis are presented in Figure 3.13. The trends of water temperatures from 2001 to 2018 in the main river can be roughly divided into three groups, primarily controlled by geographical location with a gradual increase in water temperature from the north to the south. First of all, three upstream sites in the north have distinctive patterns in water temperature. For instance, sites M1 and M2 reported 2–3 °C lower water temperature than the rest of the sites during the whole period and site M3 has a similar trend as M2 with 1 °C higher. Sites M4–M6 were closely correlated with each other, apart from the rest of the sites until 2008 when they became divergent temporarily. Later, they re-converged in 2011 and the three have started to follow the same pattern of trends as at sites M7–M15. Sites M7–M15 at mid- and downstream of the main river are roughly grouped together over the whole period, behaving similarly with some degree of variability. However, their trends became even more highly correlated since 2011.

Overall, the time-series of water temperature in the main river is characterised by a period of high water temperature in 2009, a period of low water temperature between 2010–2011, a period of rapid rise from 2013, and a period of persistent high water temperature between 2015–2017. These temporal trends in water temperature are similar to the temporal changes in daily air temperatures observed in the study area (Figure 3.10). Also, from 2011, the water temperature at sites M4–M15 became more closely correlated with each other compared to the previous state.

Conversely, time-series trends of monthly water temperature in the tributaries show little correlation between sites with more degree of variability by the river (Figure 3.14). Unlike the main river, the degree of correlation in water temperature is weak and it appears that water temperature is more locally controlled by catchment features. For example, tributary HG (T9a and T9b in Figure 3.14 (c)) has the lowest water temperature, although it flows in the southwest region. Tributaries KH (T7a and T7b in Figure 3.14 (b)) and HO (T8 in Figure 3.14 (b)) in the mid-region have the highest water temperatures because of dry weather condition in the areas.

Nevertheless, overall time-series trends of monthly water temperature in the tributaries are similar to the trends of air temperature; water temperatures were high in the late 2000s before a period of low water temperature between 2010–2011. Then

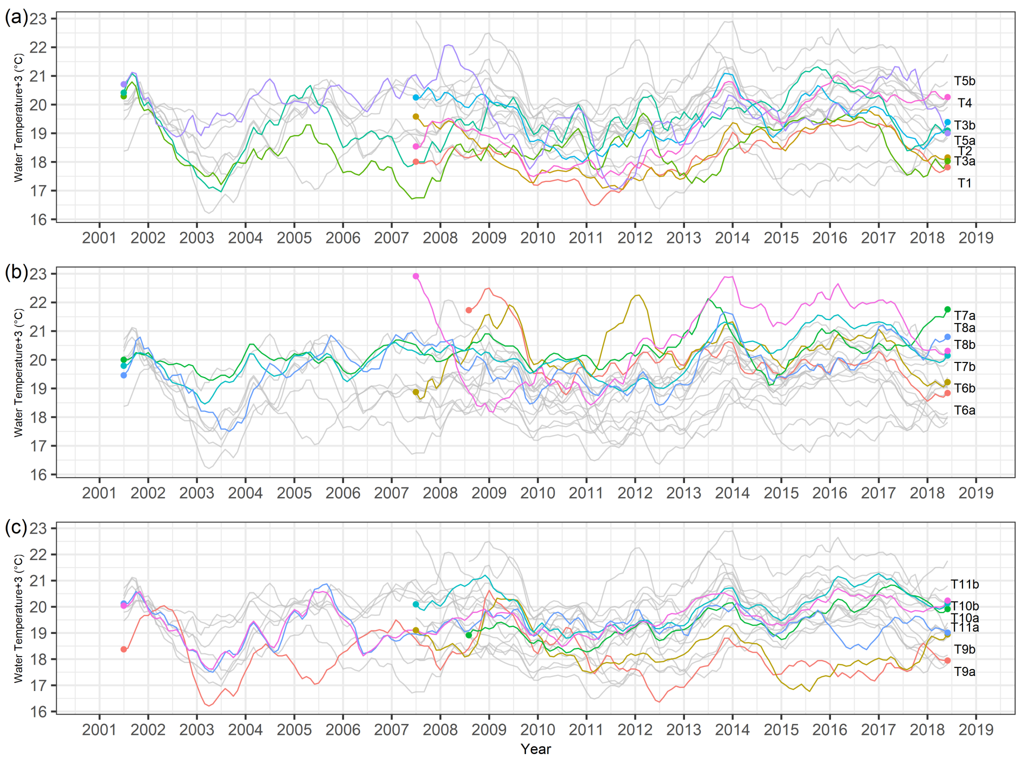


Figure 3.14: Time-series trend of monthly water temperature in the tributaries. (a) Trends in BB (T1), NS (T2), YG (T3a and T3b), BS (T4), and WE (T5a and T5b), (b) Trends in GM (T6a and T6b), KH (T7a and T7b), and HO (T8a and T8b), (c) Trends in HG (T9a and T9b), NG (T10a and T10b), and MR (T11a and T11b). Note: the starting point for each site differs. The division of (a), (b), and (c) is roughly based on geographical locations.

it rapidly increased from 2013 and remained high until the summer of 2017. A period of high water temperature persisted between 2013 and 2017 at most tributary sites, which is the same in the main river. However, during the period, some sites in the tributaries did not increase in water temperature but remained similar. For example, water temperatures in tributary HG (T9a and T9b) remained low without any big movement. Tributary GM (T6a and T6b) increased in water temperature over the period, but the extent of change was not big as those in 2011 when it was a low period of water temperature in general in the Nakdong catchment.

Overall, the time-series trends of monthly water temperature from 2001 to 2018 in the tributaries exhibit a similar trend following the changes in air temperature in the Nakdong catchment, although water temperatures in the tributaries are less correlated with each other with more variability in the trends. This means that water temperature in the tributaries is locally controlled by the catchment and catchment features are important in understanding changes in water temperature in the tributaries. Also, there is no clear change detected in the time-series trends in the tributaries, unlike the main river where water temperature started to behave in a similar way.

The observed time-series trends of monthly water temperatures in the main river and the tributaries were tested using the Mann-Kendall test to see if the trend line has a monotonic movement. The results are shown in Figure 3.15 and Table 3.3. Most sites in the rivers have upward trends and the most significant changes are observed in sites M4–M15 in the main river, which coincide with reaches of the main river affected by the barrages. Some sites in the tributaries that are adjacent to the main river also show a significant upward change (e.g. T5b, T7b, T8b, and T10b). Meanwhile, some sites in the tributaries that are not close to the main river have also upward trends. For example, T7a in tributary KH and T10a in tributary NG have a strong upward trend. They may be explained by local climatic features such as dry climate conditions or the impact of cities such as Jinju and Daegu. In contrast, the other sites display a downward trend or no trend in the time-series of monthly water temperature. Tributary HG (T9a and T9b) shows a downward trend, while M1 and M3 in the main river and T6b in tributary GM are confirmed to have no clear trend.

Based on the obtained time-series trends of monthly water temperature in the rivers, a change point at each site was sought using the Pettitt's test and the results are presented in Table 3.3. Sites M4–M15 in the main river which have an upward trend in the Mann-Kendall test are confirmed to have a change point in the observations around February or March 2013. Similarly, some sites in the tributaries (e.g. T5b, T7b, T8a, T8b, T10a, T10b, and T11b) have a change point at the same time or at around a similar time in 2012. However, these results should

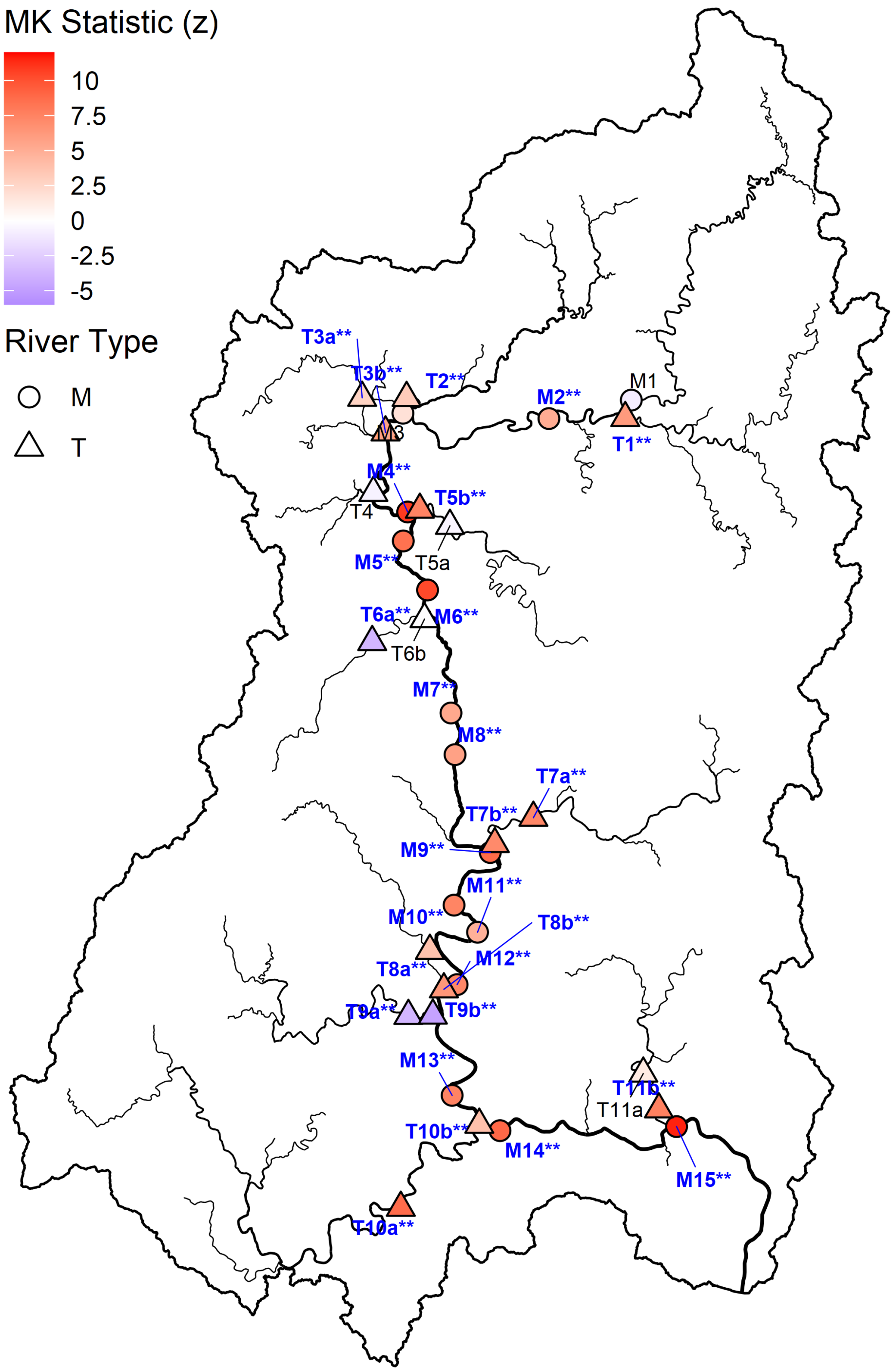


Figure 3.15: Map showing the results of the Mann-Kendall trend test for time-series trends of monthly water temperature in the study area. Sites with two asterisks in blue bold letters have $p < 0.05$. Sites in black plain letters have $p > 0.05$ in the Mann-Kendall trend test.

Table 3.3: Results of the Mann-Kendall test and the Pettitt's test for time-series trends of monthly water temperature in the main river and tributaries. (N=Number of observations)

Site	N	MK Statistic (z)	Sample estimates (S)	Var (S)	Tau	$p-value$ for MK Test	Pettitt's Statistic (U)	Probable Change point (K)	Date for K	$p-value$ for Pettitt Test
M1	204	-0.948	-925	950145.6	-0.045	0.343	5224	116	2011-02-01	0
M2	168	5.138	3747	531475	0.267	0	4577	111	2013-09-01	0
M3	132	1.873	953	258337.3	0.111	0.061	2558	91	2015-01-01	0
M4	204	10.932	10655	949851.6	0.517	0	8520	142	2013-04-01	0
M5	204	8.491	8278	950135.6	0.4	0	8587	140	2013-02-01	0
M6	204	10.389	10126	949905.3	0.491	0	8023	141	2013-03-01	0
M7	204	5.423	5287	950155	0.255	0	6060	140	2013-02-01	0
M8	204	5.808	5662	950133.6	0.274	0	6645	141	2013-03-01	0
M9	204	8.832	8610	950128.3	0.416	0	8274	140	2013-02-01	0
M10	204	7.457	7270	950123.3	0.351	0	7101	141	2013-03-01	0
M11	204	4.891	4768	950122.6	0.231	0	6076	141	2013-03-01	0
M12	132	7.338	3731	258416.3	0.432	0	4337	69	2013-03-01	0
M13	120	7.549	3329	194365.6	0.466	0	3269	43	2012-01-01	0
M14	204	8.959	8734	950158.6	0.422	0	7992	140	2013-02-01	0
M15	204	11.571	11280	950099.3	0.546	0	8254	140	2013-02-01	0
T1	132	6.228	3167	258412.6	0.366	0	3687	75	2013-09-01	0
T2	132	3.24	1648	258411.3	0.191	0.001	3017	75	2013-09-01	0
T3a	204	2.724	2656	949984.6	0.129	0.006	3680	149	2013-11-01	0
T3b	204	5.749	5604	950016.6	0.271	0	5122	126	2011-12-01	0
T4	132	-0.681	-347	258413.6	-0.04	0.496	1898	26	2009-08-01	0
T5a	204	-0.497	-485	949967	-0.024	0.619	5445	91	2009-01-01	0
T5b	132	7.532	3830	258415.3	0.443	0	4306	70	2013-04-01	0
T6a	119	-3.744	-1631	189567	-0.232	0	1554	15	2009-10-01	0
T6b	132	-0.039	-21	258391.6	-0.002	0.969	1271	119	2017-05-01	0.031
T7a	204	7.304	7120	950108	0.344	0	5487	62	2006-08-01	0
T7b	204	7.138	6959	950140	0.336	0	7677	141	2013-03-01	0
T8a	204	3.842	3746	949903.6	0.182	0	4875	142	2013-04-01	0
T8b	132	6.407	3258	258415.3	0.377	0	3541	63	2012-09-01	0
T9a	204	-3.744	-3650	950070.3	-0.177	0	5427	115	2011-01-01	0
T9b	132	-4.776	-2429	258415.3	-0.281	0	2665	87	2014-09-01	0
T10a	119	8.762	3816	189563	0.544	0	3042	55	2013-02-01	0
T10b	132	3.964	2016	258408.6	0.233	0	2410	71	2013-05-01	0
T11a	204	1.274	1243	949994	0.06	0.203	2401	79	2008-01-01	0.035
T11b	204	7.657	7465	950147	0.361	0	7084	134	2012-08-01	0

be carefully interpreted when the number of observations is small.

This time, the time-series of monthly water temperature from 2001 to 2018 in the rivers were compared by the same month each year over time to evaluate their seasonal trends using the Seasonal Mann-Kendall test. The results of the test are presented in Figure 3.16 and Table 3.4. Most sites that show seasonal upwards trends are detected in the main river along with seven sites in the tributaries. In particular, ten sites that have an upward trend in the Mann-Kendall test are found to have strong seasonal upwards trends in the Seasonal Mann-Kendall test except for M7 and M11. In the tributaries, sites with meaningful seasonal upward trends are restricted to sites that are close to the confluence of the main river, such as T5b, T7a, T7b, T8b, and T11b.

The spatio-temporal pattern of the seasonal upward trends in monthly water temperature in the rivers differs between the rivers. The main river has sites M4–M15 (except M7 and M11) that show seasonal trends, which are located in the reaches affected by the barrages and the most significant upward movements are identified in March, May, and July. In particular, sites M4–M6 experienced increases in water temperature throughout the year during the whole period with the biggest change in September. In the downstream of the main river, sites M14 and M15 show the biggest increases in monthly water temperature between May and July. This phenomenon is also detected at T11b which is located in tributary MR between M14 and M15. Contrary to the changes in the main river, the tributaries have varying patterns of the seasonal increase in water temperature throughout the year by sites. For instance, T5b shows increases in the seasonal temperature throughout the year except for February, while T8b shows increases except for May and August. T7a and T7b in tributary KH display the substantial seasonal increases between March and August. Also, T7b sees rises in January and February. Overall, few sites in the tributaries have statistically significant seasonal temperature increases when compared to the main river. Those sites in the tributaries are spatially limited to sites close to the confluence of the main river.

3.4.3 Changes in water quality

In order to understand the overall annual trends of water quality from 2001 to 2018 in the Nakdong catchment, nine water quality variables — pH, DO, BOD, COD, EC, SS, Chla, TN, and TP — were compared between the main river and the tributaries. Then, time-series trends of water quality in the main river were produced by site and mean value and change points for each variable were sought. Those change points in the time-series trends were evaluated in connection with the construction of the barrages and the hydrological system in the main river.

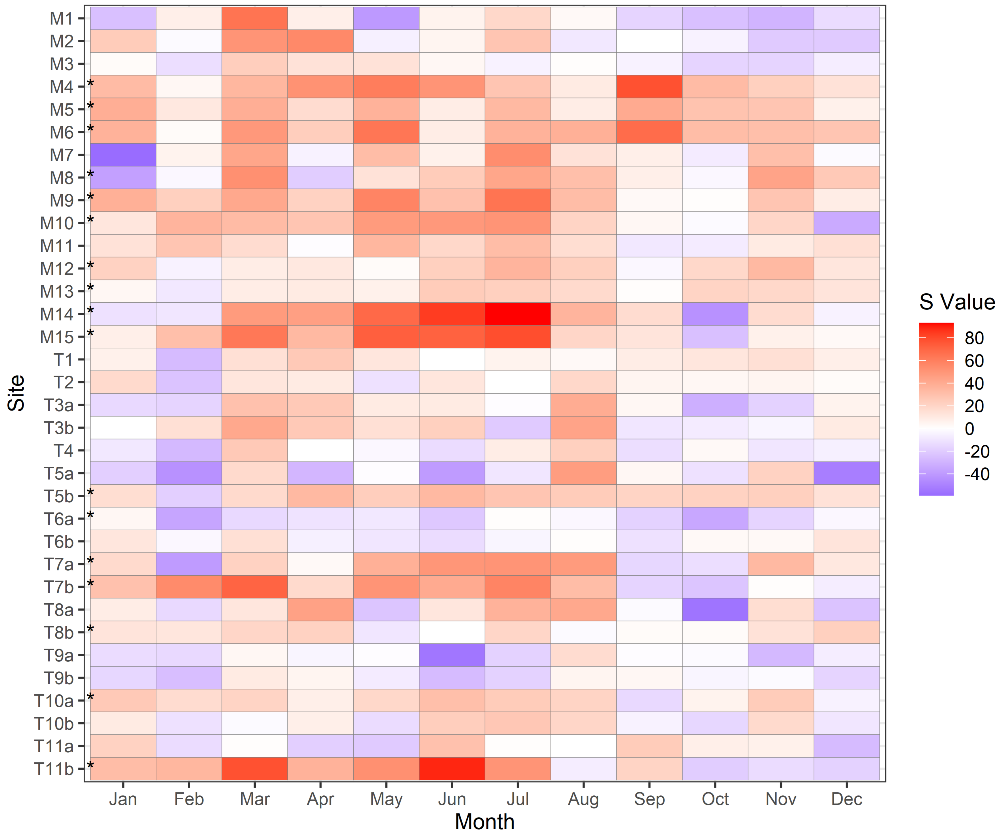


Figure 3.16: Matrix showing the results of the Seasonal Mann-Kendall test for monthly water temperature in the main river and tributaries. Sites with an asterisk mark have $p < 0.05$.

3.4.3.1 Annual trends of water quality in the main river and the tributaries

Annual means of water quality from 2001 to 2018 in the main river and the tributaries are presented in Figure 3.17. It shows that throughout the whole period, there is no absolute monotonic upward or downward trend across all the variables, but the pattern and timing of changes are different by variables and rivers.

Trends of pH in the main river and the tributaries show contrasting responses from time to time. They made a similar pathway until 2011 with a drop in 2003 recorded. From 2011, pH in the main river surpassed the tributaries and increased over 8 from 2012 and remained high until 2017. However, the tributaries did not have a rise in 2011 and remained stable at between 7-8. In 2018, the pH level in

Table 3.4: Results of the Seasonal Mann-Kendall test for monthly water temperature in the main river and tributaries

Site	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	S	Var S	$p-value$
M1	-25	8	65	8	-41	6	19	3	-17	-25	-30	-14	-43	8253.7	0.644
M2	25	-2	51	56	-6	5	28	-9	0	-5	-21	-21	101	4843.7	0.151
M3	2	-13	24	14	14	4	-5	1	-5	-17	-17	-7	-5	2509.7	0.936
M4	33	4	35	52	61	51	28	10	78	33	23	14	422	8192.7	0
M5	39	11	39	17	37	9	34	9	41	29	28	7	300	8323.3	0.001
M6	37	2	49	24	64	9	37	38	68	32	31	28	419	8160.3	0
M7	-59	6	43	-5	32	7	54	14	8	-8	31	-2	121	8335	0.189
M8	-38	-3	53	-20	14	25	43	31	8	-3	44	26	180	8296	0.049
M9	38	23	42	22	58	30	65	32	3	1	28	9	351	8287.7	0
M10	12	36	33	28	48	49	51	21	4	-2	20	-34	266	8282.7	0.004
M11	14	28	17	-1	35	19	32	16	-9	-8	10	15	168	8292	0.067
M12	22	-5	9	11	2	23	36	23	-3	19	34	12	183	2540.3	0
M13	4	-9	9	10	7	25	23	18	1	21	19	13	141	1977	0.002
M14	-12	-10	48	46	69	84	93	36	17	-44	17	-5	339	8311	0
M15	8	31	63	34	73	72	79	20	13	-25	7	3	378	8247.3	0
T1	7	-27	15	26	12	0	6	3	9	12	15	8	86	2544	0.092
T2	18	-24	12	10	-12	12	0	19	5	4	5	2	51	2547	0.322
T3a	-15	-17	30	26	10	10	-1	40	4	-32	-18	6	43	8201.7	0.643
T3b	0	15	42	26	15	23	-21	44	-10	-8	-4	10	132	8232	0.149
T4	-9	-28	26	0	-3	-14	9	23	-13	3	-10	-6	-22	2542	0.677
T5a	-19	-44	18	-29	-1	-40	-10	47	4	-12	22	-52	-116	8200.7	0.204
T5b	16	-19	18	34	24	34	28	25	21	22	23	14	240	2548	0
T6a	4	-36	-15	-11	-9	-22	1	-3	-18	-35	-17	-3	-164	1936	0
T6b	12	-3	15	-6	-10	-14	-4	1	-12	3	3	13	-2	2538.7	0.984
T7a	18	-40	22	3	38	50	51	48	-16	-13	34	11	206	8277.3	0.024
T7b	30	55	71	18	51	41	58	32	-17	-23	1	-7	310	8327.3	0.001
T8a	9	-15	12	45	-23	12	37	42	-2	-56	16	-24	53	8128.3	0.564
T8b	13	12	20	22	-10	0	20	-2	2	2	14	23	116	2550	0.023
T9a	-14	-15	4	-4	-1	-55	-18	17	-1	-2	-28	-7	-124	8280	0.176
T9b	-16	-26	10	5	-8	-27	-18	5	4	-4	-2	-17	-94	2544	0.065
T10a	26	17	21	8	19	31	25	21	-15	6	25	-5	179	1932.3	0
T10b	10	-12	-2	8	-14	24	27	20	-5	-16	18	-10	48	2550	0.352
T11a	22	-14	1	-19	-21	30	1	0	25	8	7	-27	13	8221	0.895
T11b	32	35	78	37	53	89	51	-7	21	-20	-14	-18	337	8315	0

the main river dropped to a similar level as in the tributaries.

Temporal changes in DO are different between the main river and the tributaries. The DO level in the main river was lower than in the tributaries before the mid-2000s. From 2009, the main river surpassed the tributaries in DO level and gradually increased. However, the tributaries were similar in DO level throughout the whole period without substantial changes.

BOD concentration in the main river and the tributaries is divergent over time. Both rivers recorded a fall in the concentration in 2003. Afterwards, the tributaries had a period of surges during the mid-2000s before they came down below the main river in BOD concentration. Meanwhile, since the drop in 2003, the main river also

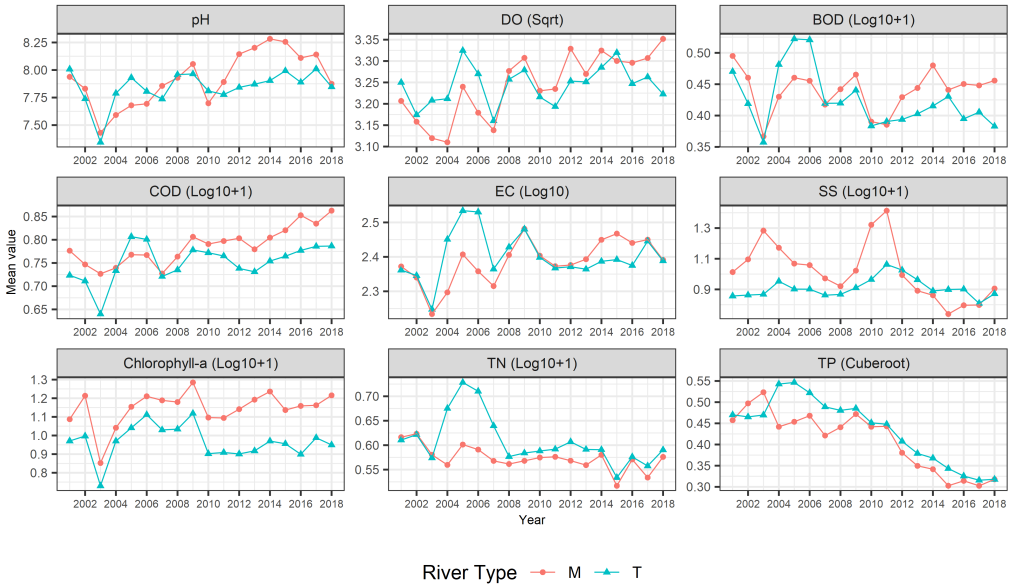


Figure 3.17: Annual means of water quality in the main river and the tributaries from 2001 to 2018. River Type M and T represent the main river and the tributaries.

rose and remained similar in BOD over time.

In COD concentration, the main river and the tributaries have an increasing trend over time, but the slope is much steeper in the main river than in the tributaries. The main river is on a gradually increasing trend during the whole period, although there was a small drop on two occasions in 2003, and 2007. The tributaries recorded a significant drop in 2003 and they remained similar with a gentle slope in the increasing trend from 2013.

Trends of EC in the main river and the tributaries are similar with no strong upward or downward trend over time. In 2003, both rivers recorded the lowest EC level and had surges between 2004 and 2009. Later, the main river remained flat until 2013 when the EC concentration started to gradually increase. The tributaries remained low and they showed a sign of an increase in 2017, although the upward trend in the tributaries was not obvious.

The level of SS between the main river and the tributaries is contrasting. Until 2012, the main river had been always higher in SS level with two peaks identified in 2003 and 2010–2011. After the second peak, the SS level in the main river significantly dropped below the level of the tributaries in 2012 and continued to be low until 2018. However, the tributaries had a small peak with a gentle rise and fall in 2010 and 2012. The peak did not form as high as in the main river and came down to the previous level of SS.

Chla concentration in the main river and the tributaries shows a very similar trajectory during the whole period with the main river having always a higher concentration. Both rivers recorded a notable drop in 2003, but rose to the previous levels and reached a peak in 2009. In 2010, they dropped by a significant amount in the concentration. Afterwards, the main river increased close to the previous level of the concentration at a gentle slope, while the tributaries remained stable. These divergent movements in the rivers from 2011 mean that the difference between the main river and the tributaries became bigger in the 2010s.

Temporal patterns of TN between the rivers are similar during the whole period except for between 2004–2007 when there was a huge peak of the nutrient concentration in the tributaries. After the peak, TN concentration in the tributaries fell to the previous level in 2008. Afterwards, both rivers remained similar over time with a drop in the concentration in 2015 and 2017.

Finally, trends of TP in the main river and the tributaries show a significant downward direction during the period since a peak in the mid-2000s. The main river hit the highest concentration in 2003, while the tributaries reached the peak between 2004 and 2005. Since then, both rivers saw a gradual decrease and the concentration in 2015 was almost half the concentration in the early 2000s.

To summarise the findings, annual trends of water quality in the main river and the tributaries from 2001 to 2018 show that there is no common sign found across all water quality variables indicating the improvement or deterioration in water quality in relation to the construction of the barrages. In the Nakdong catchment, TP concentration has seen a significant downward trend in both rivers, which suggests that phosphorus concentration in water is effectively controlled. However, the decrease in TP concentration is unlikely to be the effect of the barrages on two points. First, their downward trends were already underway before the beginning of the construction of the barrages. Second, the tributaries where the barrages were not constructed saw an improvement in TP concentration. Meanwhile, the substantial change of SS in the main river is the direct impact of the construction of the barrages. The SS level in the main river reached the highest during the period 2010–2011 and significantly dropped below the level of the tributaries from 2012. The start of the construction in the summer of 2009 in the main river would have generated disturbance and provided sediments, which would form the peak in SS. Once the barrages were completed in April 2012 and started operating, the supply and transport of sediments in the main river were very limited.

In the meantime, other variables such as pH, DO, COD, and Chla in the main river have turned to an upward trend from around 2011–2013, which roughly overlap with the construction period or the end of the construction period. However, it is difficult to conclude that these changes were triggered by the barrages with the result of the annual mean of water quality variables. These variables in the main river needed further analysis to examine their changes in time and space. In those changes, change points in mean and variance were found to understand their changes in relation to the impact of the construction of the barrages. The results of this further analysis are presented in the next section. In contrast, the tributaries have little change during the whole period in all variables except for TP concentration.

3.4.3.2 Mean changes in water quality in the main river

Time-series trends of monthly water quality from 2001 to 2018 in the main river were plotted by sites (M1–M15) and mean change points in the trends were found. These analyses show changes in the trends of water quality in space (M1–M15) with change points in time, which can be understood in connection with the construction of the barrages in the main river. Here, I mainly focused on six variables in the main river — pH, DO, COD, SS, Chla, and TP — that have exhibited remarkable changes in annual trends (Figure 3.17), while the other variables (e.g. BOD, EC, and TN) are presented in Appendix C (Figure C.1–C.3).

All sites in the main river show an increase in pH during the whole period with different responses by sites in the upstream and downstream of the main river (Figure 3.18). From a longitudinal perspective, water in the main river generally becomes more alkaline downstream with the highest value of 8.5 observed at M9. By locations, three upstream sites M1–M3, which are located upstream of the first barrage (B1), had a plateau in the trend of pH during the mid-2010s before having a drop in 2017. Sites M4–M8 have multiple change points during the whole period by an incremental increase and the latest increasing change point in the trend is detected in 2012. Sites M9–M12 have a change point in 2012 by a marginal increase, but their upward trends are relatively less obvious. Most downstream sites M13–M15 have a big jump by 0.5 in pH level between 2012–2013. These upward changes at those sites came with the reduction of the seasonal variations in the trends.

Mean changes in DO concentration in the main river show little change compared to those in pH, but notable mean change points are identified in upstream sites M4 and M6–M8 as well as downstream sites M14–M15 (Figure 3.19). Those changes in the upstream sites are detected between 2006 and 2010 before the DO level increased in 2010. In the downstream, mean change points at M14 and M15 are found in 2012. After this upward change in the trends in 2012, the seasonal variation in the DO level at sites M14–M15 changed. The other sites in the main river have remained similar in the DO level.

In the main river, an increasing trend of COD concentration is identified in all sites since the start of the observations in 2001 (Figure 3.20) and the rapid increases primarily took place in sites M4–M9 where COD concentration was getting as high as in downstream sites M13–M15. In those sites, multiple change points in the trends are recognised in 2010, 2012, 2013, and 2014 with a reduction of

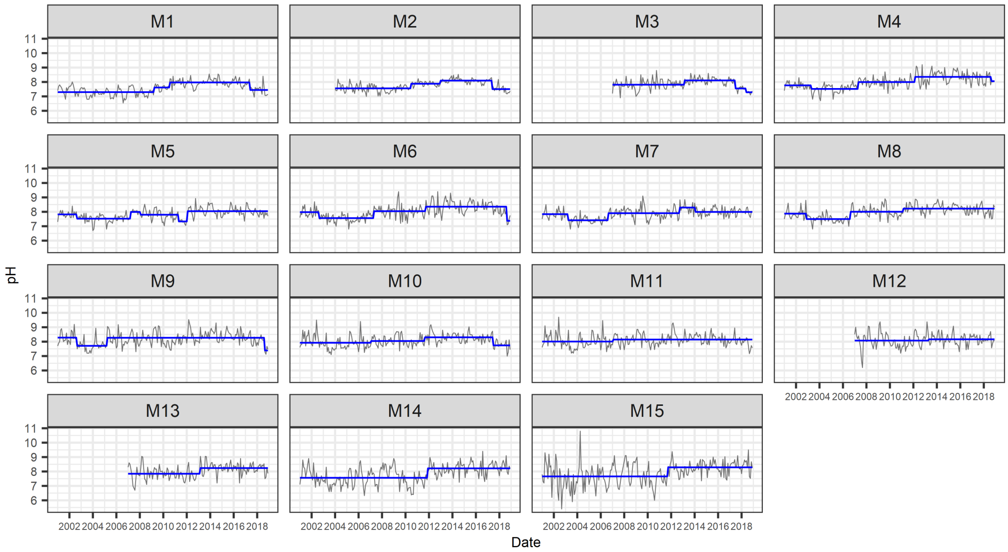


Figure 3.18: Mean changes of monthly observed pH in the main river from 2001 to 2018. The grey lines are monthly observations and the blue solid lines are mean values.

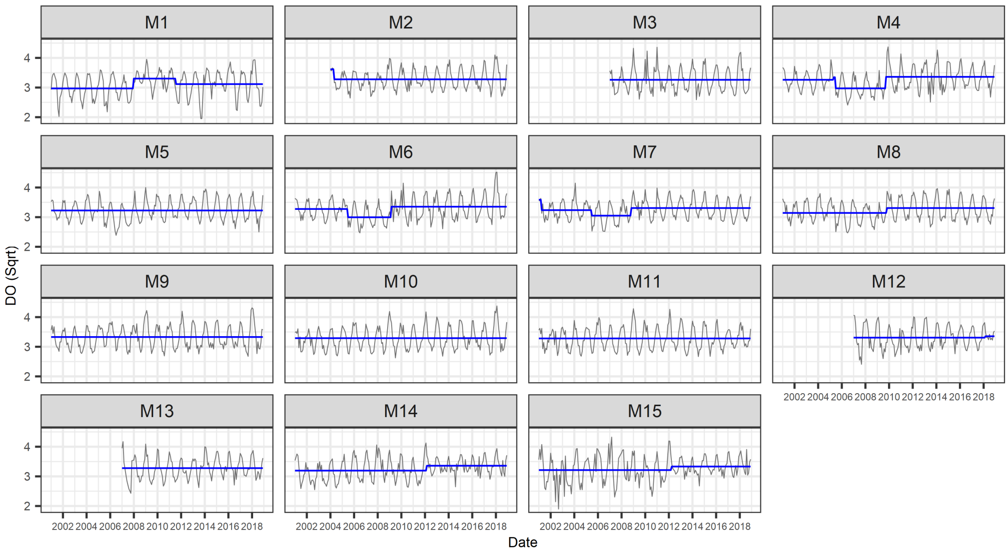


Figure 3.19: Mean changes of monthly observed dissolved oxygen (DO) in the main river from 2001 to 2018. The grey lines are monthly observations and the blue solid lines are mean values.

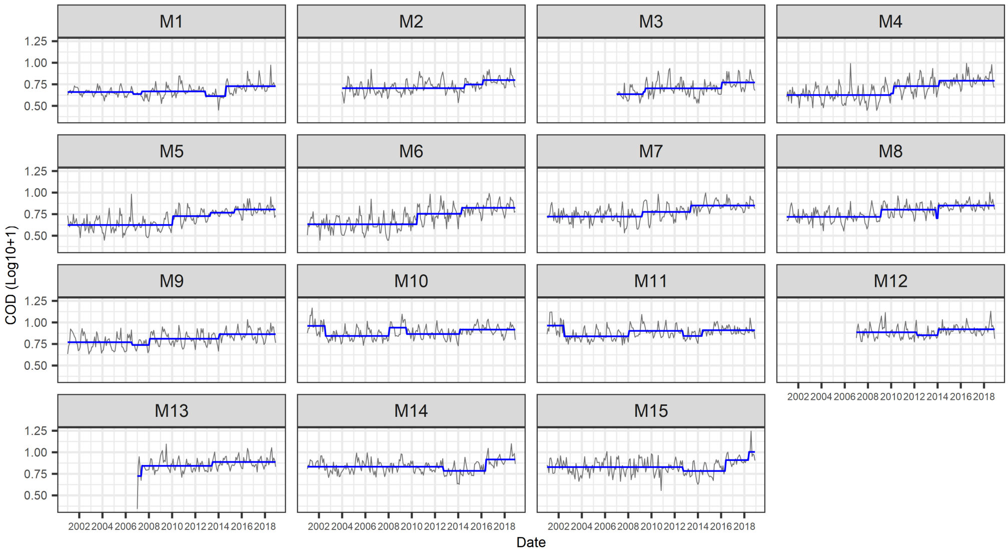


Figure 3.20: Mean changes of monthly observed chemical oxygen demand (COD) in the main river from 2001 to 2018. The grey lines are monthly observations and the blue solid lines are mean values.

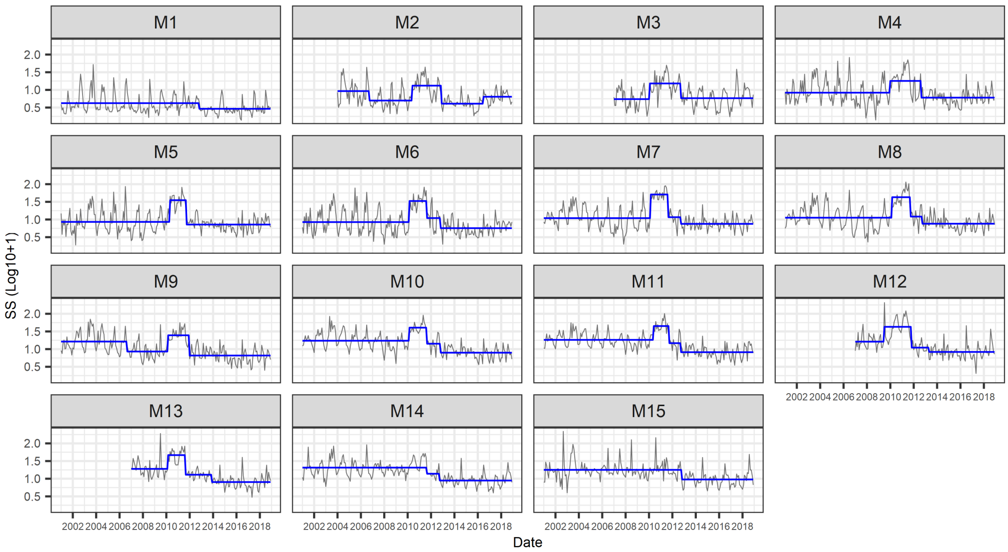


Figure 3.21: Mean changes of monthly observed suspended solids (SS) in the main river from 2001 to 2018. The grey lines are monthly observations and the blue solid lines are mean values.

seasonal changes in the concentration. In the midstream, sites M10–M12 also show incremental increases with multiple change points in the trends, but their changes are not as significant as in the upstream. Three downstream sites M13–M15 saw an increase in 2013 and 2016.

Mean changes of SS in the main river are significantly different in time and space (Figure 3.21). From a longitudinal perspective, SS values tend to be always higher towards downstream as the main river receives all sediments and water from upstream and the tributaries in the catchment. Throughout the period, the response of SS is different by the site. Most upstream site M1 has seen a relatively consistent movement in SS level with one change point detected in 2013. Sites M2 and M3 have a surge in SS during the period 2010–2013 before the sediment level returned to the previous level. Further below where the main river was influenced by the construction of the barrages, sites M4–M13 have a big spike in SS level during 2010–2011. Afterwards, they have a stepwise decrease in 2012 before the SS level further dropped down below the previous level in 2013 and remained low. These significant decreases from 2013 came with a remarkable reduction of seasonal variations in SS. However, this peak is not identified at sites M14–M15 where the mean value gradually dropped between 2011 and 2013.

The temporal pattern of Chla in the main river varies greatly between sites (Figure 3.22), although it was shown to be relatively stable during the whole period in the annual mean of the variable (Figure 3.17). Three upstream sites M1–M3, which are upstream of the first barrage (B1), show little change over the period. In contrast, sites M4–M8 have seen an increase over 1 in 2013 with a considerable change in seasonal variations. Particularly, the degree of change in Chla concentration at sites M4–M6 is significant and the concentration at those sites in the 2010s became almost as high as in downstream sites. However, Chla concentration at sites M9–M11 has been reduced after a period of high concentration during 2006–2010, but the concentration in the 2010s is still high compared to the previous level before the peak. Also, these sites have seen a reduction in seasonal variation of Chla

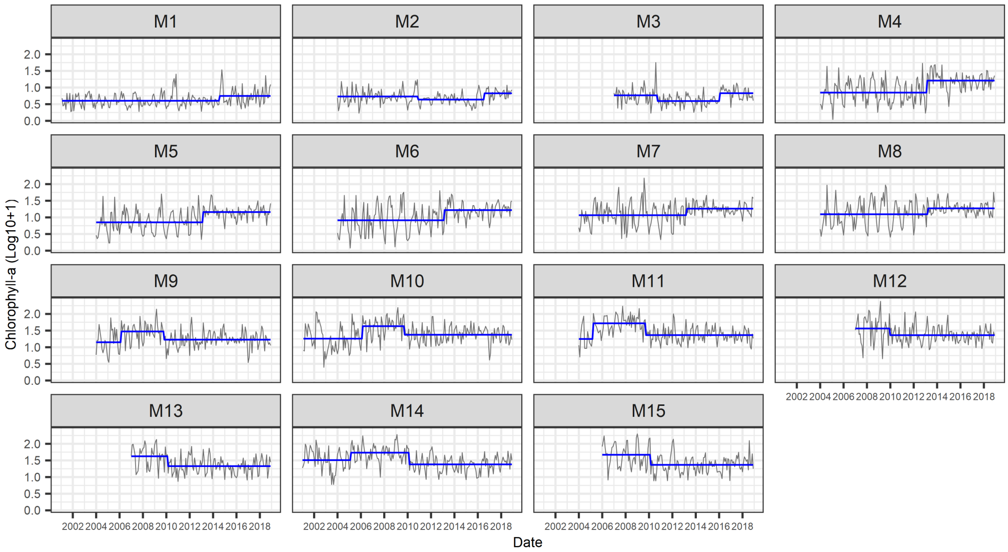


Figure 3.22: Mean changes of monthly observed Chlorophyll a (Chla) in the main river from 2001 to 2018. The grey lines are monthly observations and the blue solid lines are mean values.

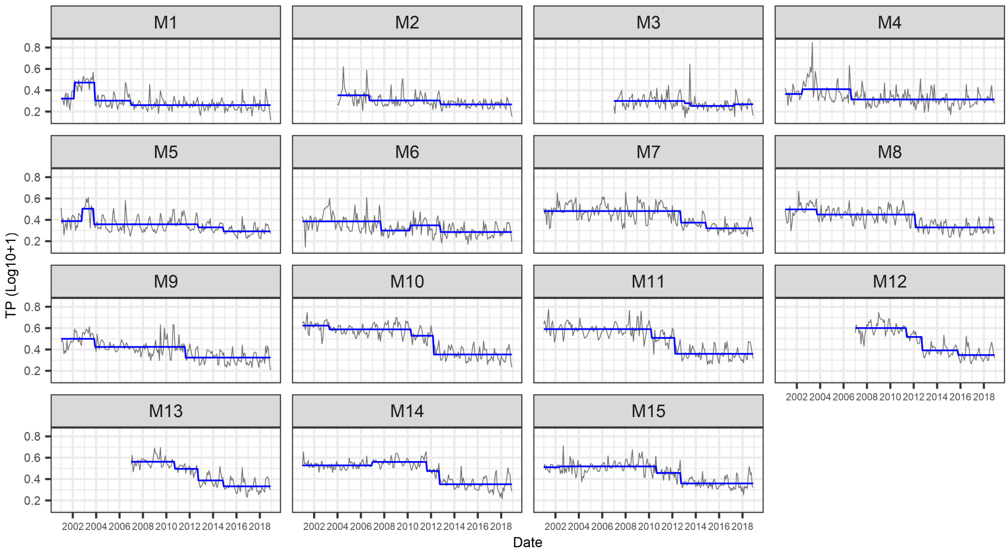


Figure 3.23: Mean changes of monthly observed total phosphorus (TP) in the main river from 2001 to 2018. The grey lines are monthly observations and the blue solid lines are mean values.

concentration since around 2012. In the downstream, sites M12–M15 have seen a decrease in the concentration of Chla with a change point detected in 2010. After the emergence of the change point in 2010, these sites saw a decline in seasonal variation. Nevertheless, Chla concentration in the downstream sites is still high in the main river.

Finally, TP in the main river has seen a gradual downturn in all sites since the early 2000s (Figure 3.23) as demonstrated in the annual mean of the variable (Figure 3.17). The most dramatic changes are observed at downstream sites M10–M15 where multiple change points in the trends are detected between 2010–2013. Besides, sites M5–M8 have also a notable drop in the trend in 2012. These significant decreases in TP concentration at the time between 2010–2013 suggest that there is a cause that drives these changes in the main river.

To summarise the spatio-temporal changes of water quality variables from 2001 to 2018 in the main river in relation to the construction of the barrages, the significant change of SS is in harmony with the construction period for the barrages. Hence, it is straightforward to say that SS is directly affected by the construction. Other changes in water quality such as increases in pH and Chla with a reduction in seasonal variation and an increase in DO appear to be closely related to the impact of the construction of the barrages as those changes are usually concentrated in sites between the barrages during the period 2010–2013.

However, a gradual rise in COD and a significant decrease in TP cannot be solely explained by the barrages as the onset of those changes began even before the construction period started in the summer of 2009. Also, it is less likely because sites M1–M3, which are located upstream of the first barrage (B1) show similar changes as in those affected by the barrages. Nevertheless, there is still a possibility that the concentration of COD would have been worsened by the barrages due to the increased retention time as some multiple change points for COD are identified during 2010–2013. For the consistent decline in TP in the main river, causal factors should be sought in human and societal efforts such as sewage treatment plants or environmental law.

3.4.4 Understanding the principal components of water quality in the Nakdong catchment

The principal components in water quality from 2001 to 2018 in the Nakdong river and its tributaries were identified using Principal Component Analysis (PCA). Identification of the principal components and the structure of observations in the dataset were essential to understanding the spatial distribution of water quality as well as utilising diatoms with water quality.

3.4.4.1 The results of PCA in the main river

In the main river, the first two principal components account for 62.6 % variance (Eigenvalue $>$ 1) (Figure 3.24 (a)). The first component is positively correlated with BOD (20.3 %), Chla (18.1 %), COD (16.0 %) and TN (11.9 %), while the second component is positively correlated with SS (25.7 %) and TP (18.3 %), but negatively correlated with DO (23.6 %) and pH (20.0 %).

The centroids of observations for each site in the main river can be roughly divided into six groups that are mainly aligned along with the first component (Figure 3.25). The order of the centroids generally follows a geographical order from upstream to downstream with the group M13–M15 located in between the group M7–M9 and the group M10–M12. The three most upstream sites consist of two separate groups of M1 and M2–M3. Site M1 is distinctly separated from sites M2–M3 which are relatively close to one another in ordination space as well as geographical space. The next group comprises sites M4–M6 in reaches of the main river which are regulated by three barrages (B1–B3). Then, sites M7–M9, controlled by three barrages (B3–B5), are correspondingly positioned on PC1 axis, and also stretched along PC2 axis. Sites M13–M15 come next along the first axis before sites M10–M12. This implies that sites M10–M12 are the most polluted among the sites in

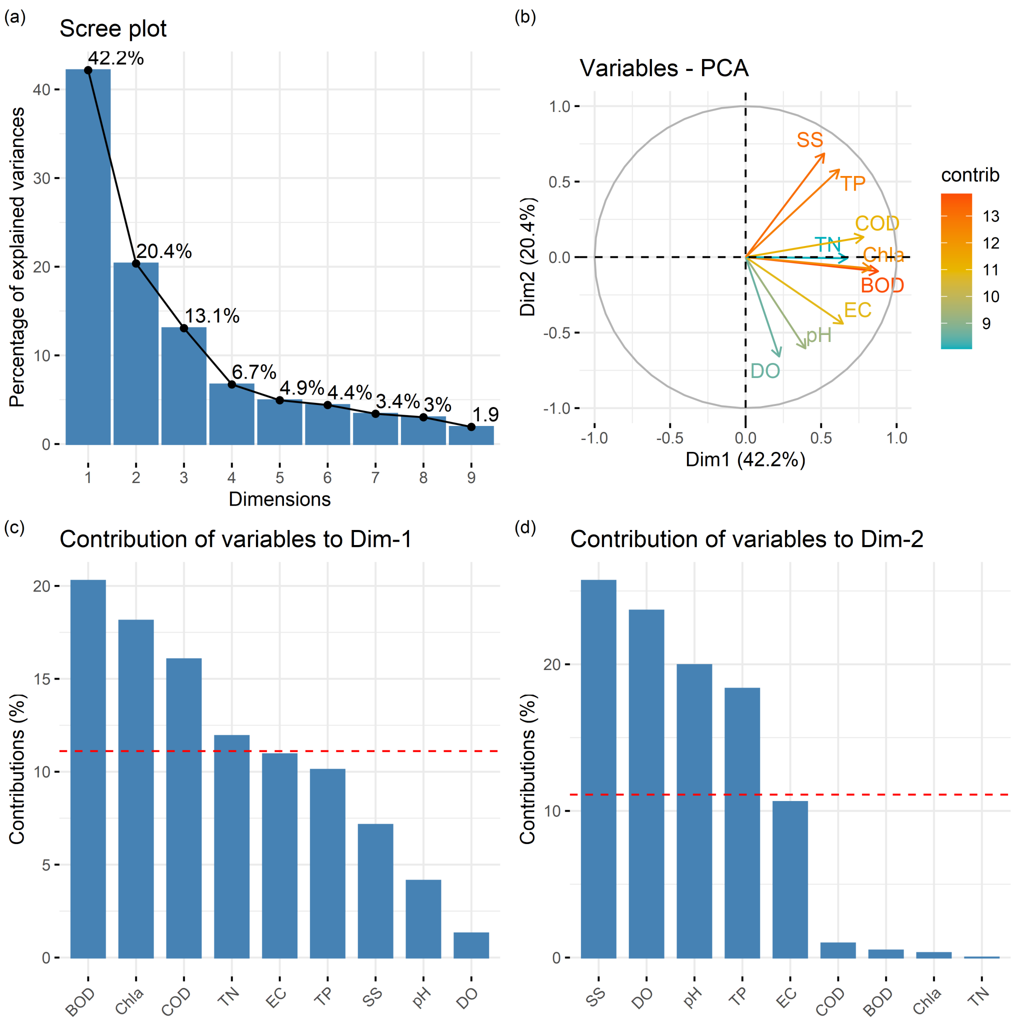


Figure 3.24: Results of Principal Component Analysis (PCA) for the main river. (a) Scree plot, (b) Variables and its contributions, (c) Contribution of variables to PCA1, (d) Contribution of variables to PCA2.

the main river. Sites M13 and M15 are nearly plotted close to the first axis, while M14 is plotted close to axis 2. This indicates that site M14 is more constrained by SS and TP due to the effect of tributary NG. In the distribution of the centroids in the main river, the effect of the second axis is limited to a few sites. For instance, M7 and M14 are positively correlated with SS and TP, affected by tributaries GM and NG that constantly supply a great amount of sand sediments and nutrients from urban areas (Gimcheon and Jinju) in their catchments. Contrarily, site M9 is positively correlated with pH and DO.

Overall, PCA results in the main river indicate that BOD, Chla, COD, and TN

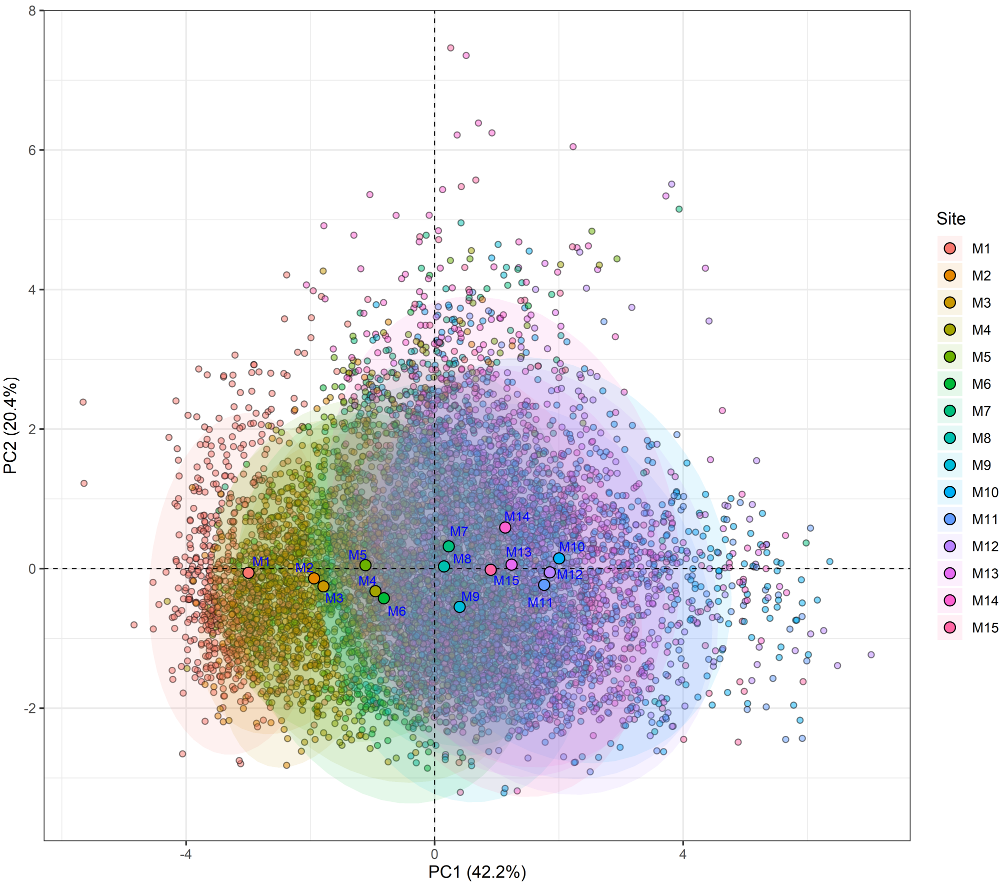


Figure 3.25: Distributions of PCA scores and centroids for the main river. Ellipses for each site are 95 % confidence level.

are dominant variables controlling the principal components in the main river from 2001 to 2018. According to the increasing gradients of these variables downstream, sites M1–M15 are distributed along the first axis and these distributions are generally followed by the geographical order. The most polluted sites in the main river are sites M10–M12, not sites M13–M15. Some sites such as M14 and M9 are partially explained by SS and TP or DO and pH along the second axis. These distributions of centroids in the main river indicate a strong longitudinal gradient in water quality with the highest pollution level at sites M10–M12.

3.4.4.2 The results of PCA in the tributaries

The same analysis was carried out in the tributaries to find the principal components in water quality data from 2001 to 2018. In the tributaries, similar variables were expected to be dominant as in the main river because most tributary sites are located in downstream of each river, below large industrial or urban areas in the catchment (see Figure 1.2 (d)). The PCA results show that the first two principal components account for 63.5 % variance (Eigenvalue > 1) (Figure 3.26 (a)). Similar to the results of the main river, the first principal component is positively correlated with BOD (19.6 %), COD (18.9 %), Chla (17.4 %) and TP (15.7 %), while the second principal component is positively correlated with SS (28.1 %), but negatively correlated with DO (31.1 %), EC (20.5 %) and TN (13.1 %).

The distribution of centroids for each site in the tributaries is mainly scattered along the first axis with a greater dispersion along the second axis than in the main river (Figure 3.27). This means that variables explained by the second axis are important in understanding water quality in the tributaries. Unlike the main river, there is no geographical order observed in the distribution at a catchment scale. However, there is a similar geographical order observed along the first axis at each tributary scale. For example, downstream site b is usually situated further right on the first axis than upstream site a. (e.g. T3a and T3b in tributary YG, T9a and T9b in tributary HG, T10a and T10b in tributary NG, and T11a and T11b in tributary MR). This structure means that the pollution level within a river increases in a downstream direction as the same in the main river. In contrast, some tributaries have the centroids of two sites in the same ordination space. For example, tributaries GM and HO have two sites (T6a and T6b, and T8a and T8b) almost overlapping in ordination space, although their geographical distance is great (15 km and 9 km). This indicates that those rivers have a weak downward gradient between the two sites.

In the distribution of centroids, observations from T7a and T7b in tributary KH are

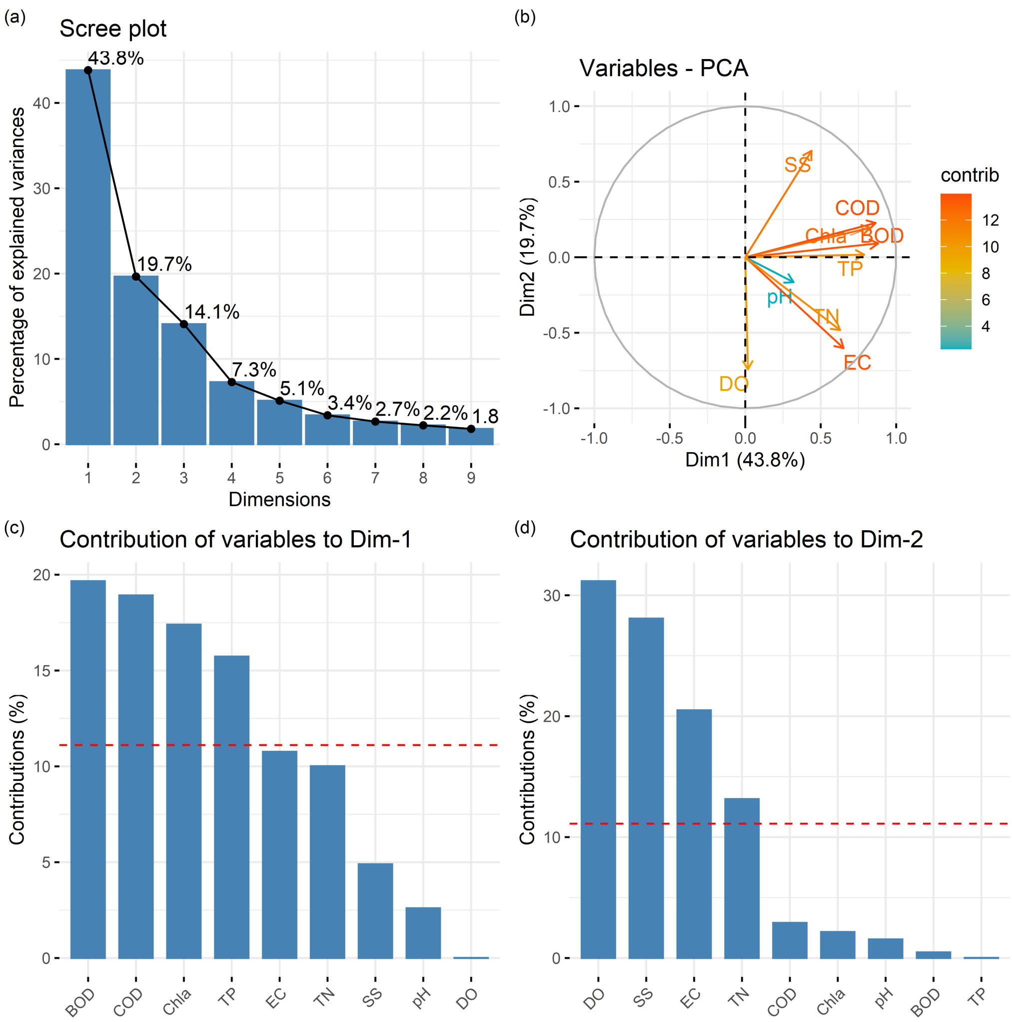


Figure 3.26: Results of Principal Component Analysis (PCA) for the tributaries. (a) Scree plot, (b) Variables and its contributions, (c) Contribution of variables to PCA1, (d) Contribution of variables to PCA2.

considerably different from the rest of the observations, characterised by a strong correlation with the first axis. This result means that tributary KH is the most polluted river in the tributaries with a high concentration of BOD, COD, Chla, and TP. This high level of pollution in tributary KH is thought to be the influence of the highly populous city Daegu and compact land use in the catchment and affects poor water quality at sites M10–M12 in the main river as seen in Figure 3.25. Contrarily, T3a and T3b in tributary YG and T9a and T9b in tributary HG are the least polluted sites, negatively correlated with the first axis. T10a and T10b in tributary NG are positively correlated with high SS, which influences the

Figure 3.27: Distributions of PCA scores and centroids for the tributaries. Ellipses for each site are 95 % confidence level.

location of M14 in the ordination space, being close to the direction of high SS in the main river.

To summarise the PCA results in the tributaries, the dominant variables are BOD, COD, Chla, and TP, which are similar to the main river. Within each river in the tributaries, a downward gradient is identified between the locations of two centroids as the same as in the main river, however, this gradient is not always obvious. This means that environmental gradients are not significantly different between two sites in some tributaries (e.g. T6a and T6b, and T8a and T8b), despite them being far in a geographical distance. In the distribution of centroids, tributary KH is the most polluted river in the tributaries with T7a and T7b segregated from the rest of the sites in the ordination space. As a result, this high level of pollution from tributary KH would contribute to the highest pollution level at sites M10–M12 in the main river. Also, a high correlation with SS in T9a and T9b explains the location of M14 in the direction of high SS.

3.5 Discussion

3.5.1 Changes to the hydrological system

Dry conditions in the Nakdong catchment were more dominant in the 2010s than in the 2000s with the recent two dry years 2015 and 2017, hitting hard the north and the south, respectively (Figure 3.4). In the 2000s, the years of 2001, 2005, and 2008 were dry prior to a dry year in 2013. The temporal pattern of dry years across the study area as a whole appears to have a two to five years of periodicity, which could be related to the effect of a warm phase of El Niño-–Southern Oscillation (ENSO) as demonstrated in the study of Lee and Julien (2017). However, Ho et al. (2016) discovered no evidence of the relationships between multiple meteorological factors affecting precipitation in South Korea and ENSO and argued that the effects of ENSO on precipitation in Korea were insignificant or negligible.

Regardless of the causes, the responses of mean discharges in the rivers are well-matched to the timing of dry periods and the tributaries tend to be more sensitive to meteorological drought than the main river (Figure 3.5–3.6). Even within the tributaries, smaller rivers (e.g. BB and BS) were more exposed to the impact of meteorological drought. As a result, the riverine ecosystem in the tributaries could not be sustained temporarily during a period of low flow. Nevertheless, the tributaries did not completely run dry due to compensation flows released from dams.

The impact of the barrages on annual mean discharge in the main river is not clear (Figure 3.5). However, it is more apparent using monthly mean discharge data that demonstrate the reduction of summer peak and the increased winter discharge in the main river since 2011 (Figure 3.7). These changes appear to be the direct effect of the barrages and extensive dredging which was carried out during the construction of the barrages. In the process, not only was river depth in the main river significantly deepened by removing sand bars and river islands, but the barrages also have caused impoundment with the increase of water level (Figure 3.8). Im et al. (2015) reported that the area of sediment bars and river islands in the Nakdong river decreased by 20.2 % and 72.7 % after the construction of the barrages. As a consequence, the main river where eight barrages were constructed has turned into a reservoir-like water body with a slow-moving flow in April 2012.

The role of the barrages in the hydrological system is different according to season. During summer, the barrages contribute to preventing the surge of a peak from forming in the main river by regulating flow, whilst they act as dams during winter by storing water and maintaining a high water level (Figure 3.9). As a result, the barrages help reduce the seasonal gap in the discharge between the monsoon and the non-monsoon period within the year. These changes in the hydrological system would have affected physical habitats in the main river. Im et al. (2020) found that there was a significant increase of permanent water area by 36.0 % and a decrease of seasonal water area and non-flooded habitat by 36.5 % and 37.4 % after the completion of the barrages in the main river.

The impoundment caused by the barrages has submerged areas that were not previously inundated during the non-monsoon period. Therefore, the impoundment would have created a steep-sloped littoral zone that is different from naturally formed littoral zones as habitats. On the other side, the disappearance of extreme peak conditions during the monsoon season in the main river would have brought stability to physical habitats. Lee et al. (2019b) and Lee et al. (2019a) discovered the appearance of vegetation and constant growth in the channel without a periodic reset of physical habitats in regulated tributary YG after the operation of a dam upstream. At daily to weekly time scales, the barrages could also cause local flooding and drought in the downstream of the barrages by directly regulating flow. Therefore, changes in the hydrological system in the main river would have affected riverine ecosystems.

However, the change of flow regime caused by the barrages is not obvious in the tributaries using annual and monthly mean discharge data (Figure 3.6 and Figure B.1–B.9) and its impact would not be as significant as in the main river. Little hydrological change in the tributaries appears to be related to the geographical distance between tributary monitoring sites and confluence to the main river as well as riverbed gradients in the tributaries. Nevertheless, some downstream areas in the tributaries very close to the confluence in distance may have been impacted in response to the increased water level.

Meanwhile, since the summer of 2017, there has probably been some degree of a change to the hydrological system following the opening of the gates at five barrages in the main river (partial opening at B1, B5, B6 and B8; full opening at B7; no change at B2–B4) (Figure 3.8). The magnitude of the change might differ by location, but decreased water level and free-flowing water at limited reaches would create diverse morphological features in the river channel that can serve as ecological habitats. In a study conducted in the Geum river in the mid-west region of South Korea, Ock et al. (2020) discovered the restoration of morphological features in the river channel such as river islands and sand bars at the downstream of the Sejong Barrage following the opening of gates.

3.5.2 The relation between water and air temperatures

The three types of daily air temperatures in the study area have all upwards trends with a slope of around 0.03 °C per year on average during a period of 18 years from 2001 to 2018 (Figure 3.10). These results are in line with the findings of Lee et al. (2011) in terms of having the same direction of movement over time. Also, the comparison of monthly air temperature between weather stations in the study area shows that the pattern is spatially well-correlated with a north-south divide at the broad scale and the monthly maximum air temperature is more correlated than the other two types of temperatures (Figure 3.11). These results mean that latitude is the main factor at the broad scale that strongly controls air temperature in the catchment. Although there is a weak geographical correlation among the stations in the monthly minimum temperature with weather stations DAEG and MIRY being the least similar, air temperature is still controlled by latitude.

The overall trajectories of water temperature observed in the main river and the tributaries (Figure 3.13 and 3.14) are similar to the time-series trend of daily air temperatures by sharing high and low periods in the trends at similar timings. These results are in agreement with those of Yu et al. (2014) that water temperature and air temperature in the study area had high Pearson correlation coefficients (0.71, $p < 0.01$) with some delay in the appearance of a peak in daily water temperature following the highest air temperature. There are also great variations in the trends of water temperature at some sites and this means that water temperature is strongly influenced by local factors. For example, trajectories of water temperatures at sites M1–M3 are considerably different from those at other sites in the main river and sites in the tributaries have less correlation with each other than in the main river. T9a and T9b in tributary HG have the lowest water temperature among the tributaries, despite them being in the south. These examples at those sites suggest that water temperature can be strongly influenced by other environmental variables such as elevation or catchment characteristics (Webb and Nobilis, 1995) instead of being controlled by latitude such as a north-south divide observed in the distribution of air temperature in the Nakdong catchment.

It appears that the barrages have played a vital role in bringing a change point to water temperature trends at most sites in the main river. The results of the Mann-Kendall test and the Pettitt's test that most sites (e.g. M4–M15) in the main river have a change point falling at around 2012–2013 (Figure 3.15 and Table 3.3) support this interpretation. Particularly, the changes at sites M4–M6 are notable that they behaved independently as a separate group before they followed the trajectories of the other downstream sites in the main river. The fact that the changes did not occur in the tributaries where there was no construction suggests that this pattern is not just a coincidence. This evidence underpins the effect of the barrages on the change in water temperature in the main river.

The barrages would have contributed to the increase in water temperature by causing river impoundment. Several studies (Webb and Nobilis, 1995; Webb and Walling, 1996; Webb, 1996) reported that river impoundment may increase water temperature through the impact on thermal capacity. For example, in the Yangtze river and its tributaries in China where the Three Gorges Dam has created reservoirs with the impoundment, Long et al. (2016) found an increase of minimum water temperature by 1.5 °C as well as lags in the heating and cooling process in monthly water temperature patterns. In the current study, it is not possible to discuss how the thermal system in the Nakdong river has changed in detail such as the extent of the increase or lags in the process given the limited data and the results. However, similar changes would probably have occurred due to impoundment in the main river. Besides, the barrages would have indirectly contributed to the increase in water temperature by increasing the retention time of water. In a lab experiment, Noh et al. (2015) demonstrated a significant increase in water temperature in surface water with increasing retention time in flow. Consequently, on top of the effect of the barrages, meteorological droughts in 2015 and 2017 in the catchment would have intensified the heating process more with limited precipitation and dry conditions.

Similarly, the results of seasonal trends of water temperature in the rivers (Figure 3.16) continue to support the impact of the barrages on water temperature. Notable increases are particularly observed in regulated reaches of the main river and its adjacent sites at the tributaries. For example, sites M4–M6 and T5b have upward trends during spring and autumn and these seasonal changes might be evidence of lags in the heating and cooling process in the main river following impoundment as was found in the Three Gorges Dam case study in China (Long et al., 2016).

To summarise, with water temperature affected by air temperature in the Nakdong catchment, the time-series trends of water temperature in the main river have a change point during 2012–2013, which is at the beginning of the operation of the barrages. At this time, water temperature at the regulated reaches of the main river started to behave in the same way. This phenomenon is not detected in the tributaries. This is evidence that the barrages have significant impacts on water temperature in the main river through impoundment and the increases in water volume and retention time. However, the impacts on the tributaries are very limited at some sites close to the main river. Accordingly, the increase in water temperature in the main river would have directly and indirectly affected water quality as well as diatoms in the main river.

3.5.3 Changes in water quality over time

Some water–quality variables appear to be closely related to extreme hydrological events such as flooding or drought as observations in some variables are particularly different below the average (Figure 3.17). For example, the year of 2003 which saw one of the biggest flooding events in the meteorological history of Korea has considerably low values for pH, BOD, COD, EC, and Chla. Although there is no discharge data for the year of 2003 in this study, but it is thought to be the extreme hydrological conditions in the river due to the highest summer precipitation. In the main river, Park et al. (2002) found that the lowest Chla concentration usually appears after the monsoonal precipitation. In the case of drought conditions like the year of 2015 and 2017, the main river and the tributaries have an increase in those variables related to flooding such as pH, BOD, and Chla, but have a drop in TN and SS. These phenomena are supported by the four ecological principles (Rolls et al., 2012); low flows can affect physical aquatic habitat, habitat conditions and water quality, sources and exchange of material and energy, and connectivity and diversity of habitat in riverine ecosystems. In addition, a lack of rainfall in a catchment would prevent runoff from generating, thus reducing the influx of pollutants to the river network system. The drop in SS is evidence of little runoff in the catchment. Jung and Kim (2017) revealed a higher Chla concentration in the dry year of 2015 than in the non-dry year of 2016 in the Nakdong river. This result is also similar to the findings (Jeong et al., 2011; Jung et al., 2014) that summer rainfall and dam discharge have a negative relationship with concentration of Chla in the Nakdong river and the Han river in South Korea. Jung et al. (2014) found that hydrodynamics such as discharge are more influential than the availability of nutrients for phytoplankton communities in the lower Han river.

During the whole period, some parameters such as DO, BOD, EC, Chla, TN, and TP in the 2010s are improved compared to those in the 2000s (Figure 3.17). Since around 2011, however, the rate of positive changes has significantly slowed down and some variables like pH, BOD, COD, and Chla in the main river have begun to rise again. These changes in direction imply the existence of changes to the catchment.

The recognition of mean changes points in the observations from the main river confirms that there are change points and one of which appears to be strongly related to the construction of the barrages in the main river, supported by the results of five variables (Figure 3.18–3.22). Parameters like pH, DO, COD, and Chla have seen a gradual increase with multiple change points including the one related to the barrages during 2012–2013. Also, SS in the main river has seen a significant decline after the peak during 2010–2011, which coincides with the construction period. Where those changes were identified are mainly concentrated at sites M4–M15 that would have been affected by the construction of the barrages in the main river and the extent of those changes is bigger in the upstream of the main river than in mid- and downstream of the main river. For example, sites M4–M8 have been particularly negatively affected by the increases of pH, COD, and Chla in 2012–2013, while other sites in the mid- and downstream have changed relatively little in those variables. In the meantime, TP in the main river has seen a consistent decrease since the early 2000s (Figure 3.23), but this downward trend is not the effect of the barrages because the trend was already underway before the construction began and this downward trend is also identified in the tributaries where the barrages were not constructed (TP in Figure 3.17).

One of the most notable changes observed among the increasing variables is pH which started to go up from 2012 until it came down in 2017 (Figure 3.18). In general, pH in water can increase because of photosynthesis activity (Yu et al., 2014) or stratification of water created by dams (Wang et al., 2015; Jung and Kim, 2017) and it is not a coincidence that this phenomenon is particularly significant at sites M4–M8 and M13–M15 where barrages have been installed. Control of the flow by the barrages results in slowing down the flow and increasing the retention time of water (Noh et al., 2015), which then allows more time for algae to grow through photosynthesis. The increases of Chla and DO (Figure 3.22 and 3.19) at these sites at the same time support this interpretation.

Furthermore, stratification of water in reservoir-like water bodies caused by the barrages may play a vital role in increasing pH levels in the main river. Meteorological droughts in 2015 and 2017 may also have contributed to the increase. Wang et al. (2015) found polarisation of pH in stratified water above dams with more acid conditions in the hypolimnion, and more alkaline conditions in the surface water because of more active photosynthesis in slow-flowing water. Likewise, Jung et al. (2016) and Jung and Kim (2017) found similar results with a high correlation between Chla and pH in the Nakdong river. Therefore, it is reasonable to conclude that the construction of the barrages has contributed to the increase of pH, Chla, and DO at sites M4–M8, and M13–M15 in the main river. The decline in pH from 2017 may be understood by the opening of gates at the five barrages.

Meanwhile, despite sites M1–M3 not being affected by the construction of the barrages, they also show an increase in pH from 2010 to 2016 (Figure 3.18). This increase might be related to an increase in photosynthesis due to an increase in light availability under dominant dry conditions in the 2010s or the effect of other construction works that happened to disturb the flow. More details on the construction are to be addressed below in connection with the increase in SS (see Figure 3.28 for locations of the construction).

The SS is more directly influenced by the construction and operation of the barrages with significant changes between 2010–2012 (Figure 3.21). In the pre-construction state until 2010, SS showed a big seasonal variation, increasing with summer precipitation and high discharge in all sites. During 2010–2011, the construction of the barrages significantly contributed to a surge in the level of solid particles at sites M4–M15 by supplying sedimentation. The effect of the construction on SS is found in previous studies. Hwang et al. (2013) reported an extremely high level of SS during the construction period in one site in the main river in comparison with an adjacent site in tributary KH. Similarly, Yu et al. (2014) reported that turbidity during 2010–2011 (the construction period) was 4.1 times turbidity during 2012–2013 (the post constuction period) in the Nakdong river. Once the construction was completed in April 2012, the SS level in the main river significantly decreased in 2012 and dropped further below the pre-construction level in 2013 in a phased manner.

Furthermore, the findings of the mean change points in the trends of SS provide an insight into the exact starting point of the construction. The administrative reports (Kim, 2009; The FMRRP Committee, 2014) stated that the construction of the barrages kicked off in the summer of 2009 and ended in April 2012. However, these dates do not necessarily mean the exact beginning or end of the construction in the river in reality. Figure 3.21 shows that the pattern of SS in 2009 is similar to that in the previous years in the pre-construction period with seasonal changes. This pattern in 2009 suggests that there is a chance that the construction had not kicked off or affected SS yet in the main river. In 2012, the peak of SS was lower than those in the construction period, but still far higher than in the later years in the post-construction period. This gradual decrease in SS in 2012 suggests that although the barrages were completed and came into operation in April 2012 (Figure 3.8 and 3.9), flow in the main river was less regulated. Since 2013, SS in the main river is greatly reduced with little seasonal variation because of the barrages in full operation. From these patterns, the onset of the actual construction in the main river can be inferred to be no earlier than 2010 and the barrages started operating in April 2012 in a phased manner.

Sites M1–M3 also saw a rise in SS during 2010–2012 (Figure 3.21) and this is well-understood with other types of construction at the sites. For example, the increase in SS at M2 and M3 during 2010–2012 appears to be related to the construction of four weirs in the main channel that took place from 2010 to 2013 (Yecheon News, 2009) (see Figure 3.28 (c)–(e)). The construction caused an increase in SS level like the construction of the barrages, but the SS returned to the previous level in 2013 after the completion of the construction. This is thought to be the different role of weirs which is to raise water level until the water can naturally flow over, while barrages can hold up water using gates (Scott, 1991). Site M1 saw a marginal rise in SS during 2011, affected by the construction of a bridge which was underway nearby during 2010–2011 (Kyongbuk Daily News, 2014) (see Figure 3.28 (b)). However, its impact seems not to be as strong as those at sites M2–M3 because of the local hydrological system. Site M1 is situated in the lower reservoir of the Andong Dam that has been used to generate hydroelectric energy. Therefore, the reservoir is dominated by stagnant to slow-flowing water, which is not in favour of solid particles being transported.

A dramatic improvement in TP concentration in the main river (Figure 3.23) and the tributaries cannot be linked with the construction of the barrages in the main river because TP concentration has been improved at a similar rate in the tributaries (Figure 3.17) where barrages were not constructed. Instead, it is more reasonable to understand this with efforts in environmental policy. The South Korean government made efforts to curb the influx of phosphorus in the rivers such as extensive investment in sewage treatment plants across the study area (Figure 3.29 and 3.30) and the introduction of a stricter criterion for phosphorus concen-

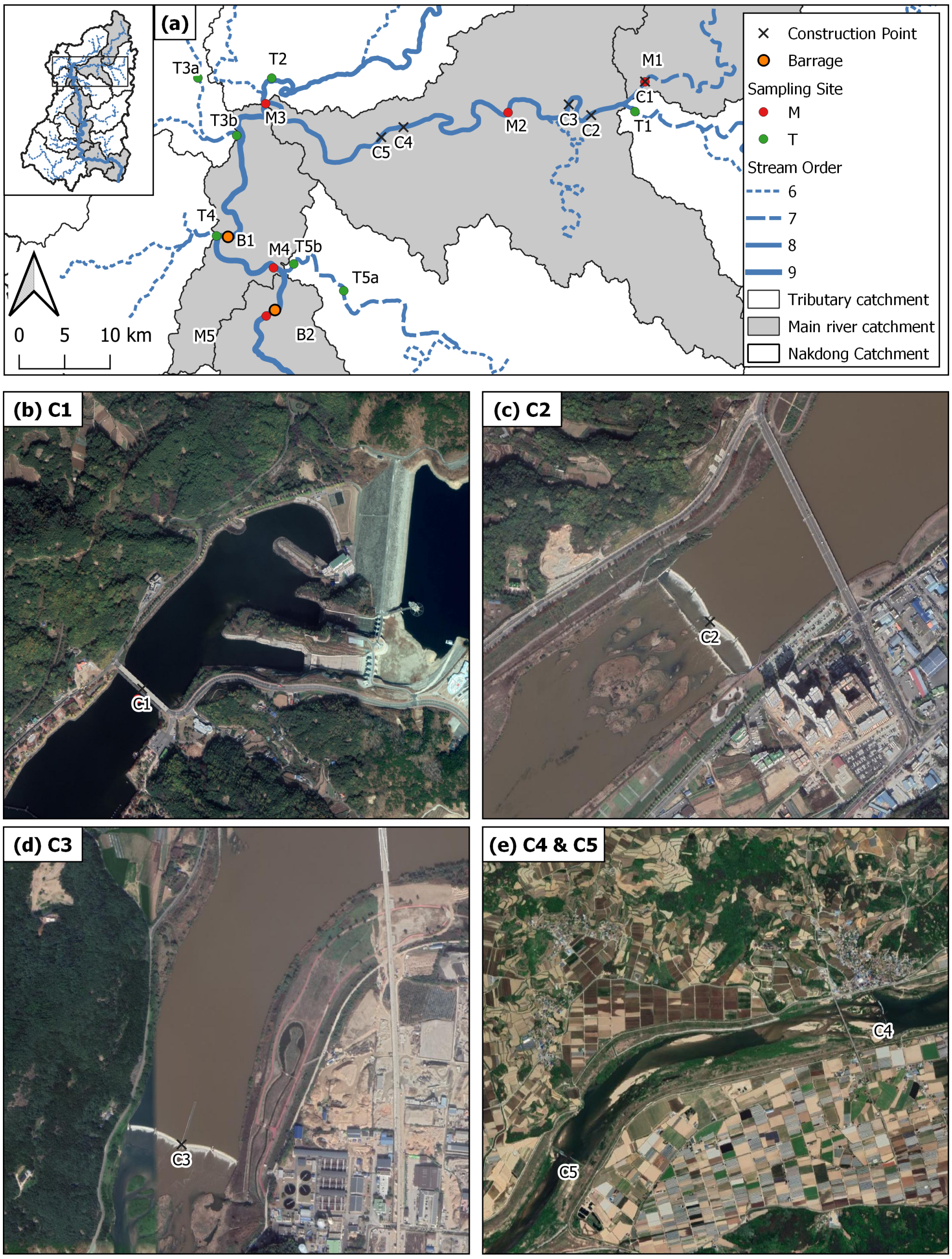


Figure 3.28: Location of other construction works carried out in the upstream of the Nakdong river. (a) Map showing the location of construction sites (C1–C5) and sampling sites in the Nakdong river, (b) A bridge (C1) constructed next to site M1, C1 is located in the lower reservoir of the Andong Dam, which is the large concrete structure on the right side. (c) Weir (C2) in the upstream of M2, (d) Weir (C3) in the upstream of M2, (e) Two weirs (C4 and C5) in the upstream of M3. All satellite images were obtained from Google Earth accessed via QGIS.

tration* in water released from treatment facilities that came into effect from 2012 (The Korean Law Information Center, 2021).

Various studies (Jung and Kim, 2017; Kim et al., 2021) indicate that TP is responsible for the growth of Chla in the Nakdong river. However, despite the nutrient being considerably reduced in the river, the increase of Chla in the main river and a similar level of Chla in the tributaries suggest that there is a role of other variables in the growth of Chla, which was discussed regarding the increasing trends in pH, DO, and Chla at sites M4–M8 and M13–M15 after the construction of the barrages.

Finally, the increase in COD concentration is observed at all sites in the main river (Figure 3.20) and mean change points for the increase are identified between 2010–2014 along with changes in seasonal variation at sites M4–M15. However, it is not clear that the barrages triggered those changes as the increase is also identified in upstream sites M1–M3. Nevertheless, the increase in COD concentration appears to be related to the barrages because of the change to slow-flowing water conditions in the main river, formed by the barrages. Tang et al. (2019) revealed that hydrological conditions such as increasing flow velocity promote COD degradation in water.

To summarise the change in water quality in the Nakdong catchment with the construction of the barrages between 2001–2018, it is not apparent to recognise the effect of the barrages on water quality using the annual mean data in the main river and the tributaries. However, the results of mean change points in the time-series trends by sites demonstrate increasing trends at the time of the construction or the operation of the barrages. Parameters like pH, DO, Chla, and COD have significantly increased in the main river at sites affected by the barrages with more dramatic changes in the up- and midstream sites (e.g. M4–M8) than in the downstream sites. Contrarily, SS has experienced a significant decline after

*The upper limit of phosphorus concentration was universally 4 mg/L in rivers in Korea before the first day of 2012 when the limits became stricter with 0.2–2 mg/L depending on areas. To meet the criterion, the tertiary sewage treatment process for TP has been introduced in the plants.

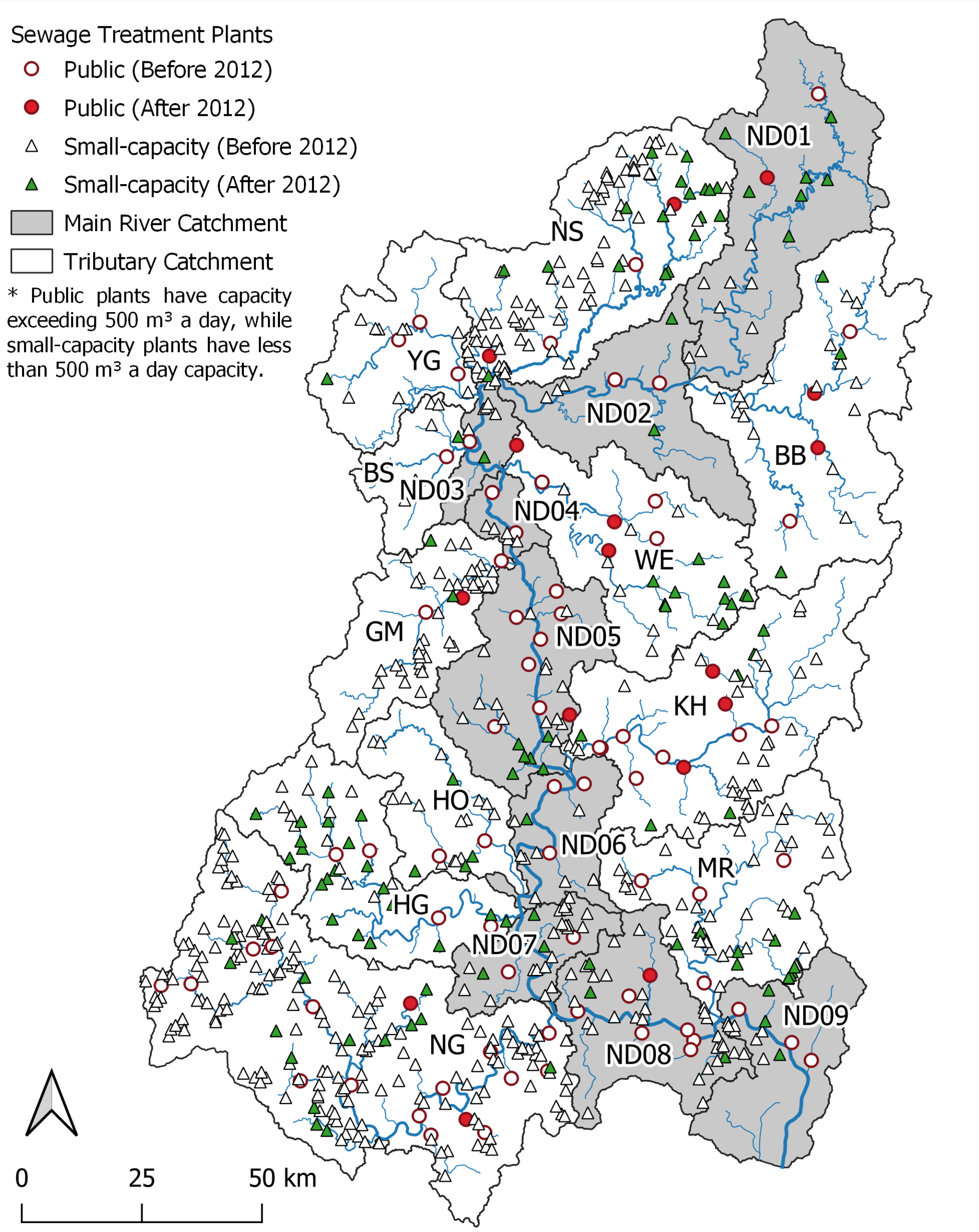


Figure 3.29: Location of sewage treatment plants in the Nakdong catchment. After 2012, there are additions of 16 public sewage plants (capacity per day > 500 m^3) and 120 small capacity plants (capacity per day < 500 m^3) across the Nakdong catchment along with the introduction of a stricter criterion for phosphorus to tackle nutrient enrichment in the rivers. As of 2018, there are 89 public sewage plants and 649 small capacity plants in operation. This map was generated using data from the Korea Environment Corporation (2020) in QGIS.

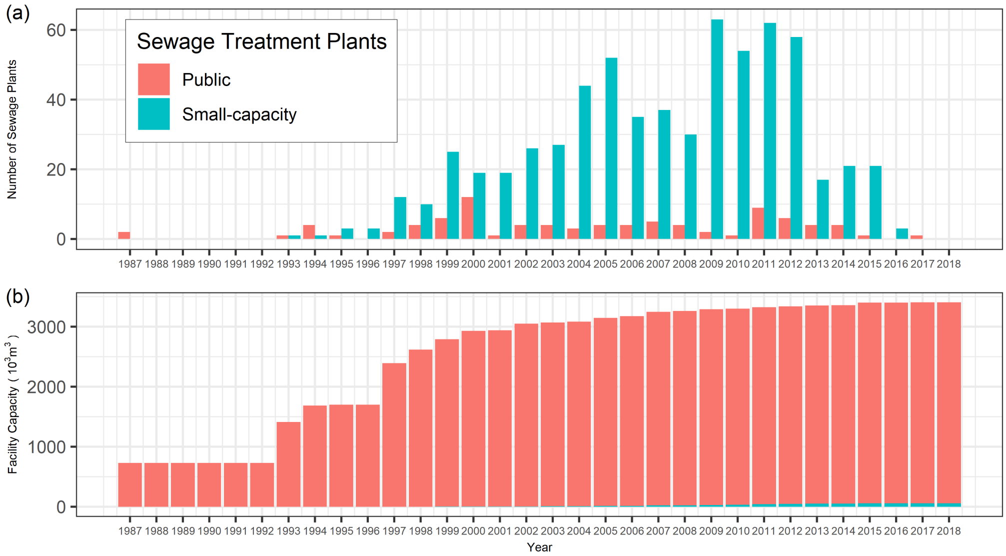


Figure 3.30: The number of new sewage treatment plants per year and their accumulated capacities in the Nakdong catchment. (a) The number of new sewage treatment plants per year, (b) Accumulated capacities of sewage plants. These graphs were generated using data from the Korea Environment Corporation (2020).

a peak during the construction period. Lastly, TP concentration in the rivers has been on the decline due to extensive efforts such as the installation of sewage treatment plants and the enforcement of the stricter criterion on phosphorus in the released water.

These findings contradict the results of Lee et al. (2019c) that looked at changes in water quality in the Geum river after the construction of barrages as part of the FMRRP. Lee et al. (2019c) made a direct comparison of four water quality variables — BOD, COD, TP, and Chla — between the pre-construction year of 2009 and the post-construction year of 2012 and concluded that downstream sites see a significant improvement in water quality after the construction of the barrages, and upstream sites see no discernible changes. However, this result is difficult to accept since they did not take into account the effects of other external factors like climate, hydrology, and other human efforts like the sewage plants and the legislation of the new law.

3.5.4 Implications of PCA results

The PCA results show that BOD, COD, and Chla are dominant variables controlling the principal components with TN in the main river and TP in the tributaries (Figure 3.24 and 3.26). These findings reaffirm the results of previous studies (Jung et al., 2016; Jung and Kim, 2017) that nutrients and organic matter are the main factors that explain variance most in the Nakdong river. Dominant variables in the PC2 are DO and SS, correlating negatively with each other in ordination space. The contribution of the two variables to the PC2 is bigger in the tributaries (31.1 % and 28.1 %) than in the main river (23.6 % and 25.7 %). It is because the tributaries are unregulated and locally controlled with variability in the catchments, while the main river is regulated by the barrages.

The difference in those five dominant variables determines the position of centroids for each site in the main river and the tributaries in PCA space (Figure 3.25 and 3.27). In the main river, sites from M1 to M15 are mainly aligned in geographical order following the increasing trends in BOD, COD, Chla, and TN along the first axis. Although sites M10–M12 come last in the axis after sites M13–M15, the alignment of all sites in the main river reveals a downward gradient at the catchment scale. This structure can be understood as a conventional view of rivers (e.g. the River Continuum Concept, Vannote et al. 1980) that sees rivers as having continuous gradients of physical variables.

On the other hand, the alignment of sites M1–M15 in the main river can be viewed as the division of six groups at a reach scale and this formation can be regarded as having downwards arrays of patches, controlled by different factors such as hydrogeomorphic processes (e.g. the Riverine Ecosystem Synthesis (RES), Thorp et al. 2006). For instance, the division among M1, the group M2–M3, and M4–M6 can be shaped by different hydrological conditions; M1 in the lower reservoir has slow-flowing water, sites M2 and M3 have free-flowing water, whilst M4–M6 have a slow-flowing water in the regulated reaches between three barrages (B1–

B3). In addition, the impacts of the tributaries may affect this structure. The division between the group M4–M6 and M7–M9 can be explained by the impacts of tributary GM and the division between the group M7–M9 and the group M10–M12 by tributary KH.

In the tributaries, a downward gradient along the first axis is also identified in some rivers (Figure 3.27). For example, tributaries YG, KH, NG, and MR have two sites aligned in geographical order from the left to the right of PCA space (e.g. T3a and T3b, T7a and T7b, T10a and T10b, and T11a and T11b). However, some rivers have two monitoring sites overlapping each other. For instance, T6a and T6b, and T8a and T8b are superimposed on each other in each river. This means that environmental gradients between the two sites are not big. On the other hand, a patchy structure is not identifiable in the tributaries due to the insufficient number of sites included along each river but it is thought that a similar pattern would appear here, underlain by their hydrogeomorphic processes.

In understanding the distribution of centroids in PCA space in the main river, the contribution of the tributaries is paramount and should be carefully considered. In particular, the location of the group M10–M12 is on the far right side of PCA space (Figure 3.25), characterised by the highest pollution level. This is because of the significant amount of pollution from tributary KH (see T7a and T7b in Figure 3.27). In the Korean literature, tributary KH was selected as one of the rivers that require more actions and measures to tackle water pollution as a top priority in the Nakdong catchment (Jung et al., 2020). Also, other tributaries make an impact on water quality at some sites in the main river. M14 is positively correlated with SS and TP in PCA space due to the impact of tributary NG.

It is important to understand these structures of water quality in the rivers in the Nakdong catchment as a basis because they can ultimately provide physical habitats and determine the distribution of diatoms in the rivers. These patterns at the catchment and reach scales will be brought in to understand the patterns of diatom groups in the rivers in Chapter 6.

3.6 Conclusions

This chapter has answered research question 1 by examining the spatio-temporal pattern of water quality in the Nakdong river and its tributaries in relation to changes in climate and hydrological systems in the Nakdong catchment between 2001 to 2018.

First, it has looked at the changes in the hydrological system in the rivers in connection with precipitation and the construction of the barrages. It has also examined the change in water temperature in comparison with the change in air temperature and investigated the impact of the barrages on water temperature. Furthermore, this chapter has examined the spatio-temporal pattern of water quality and scrutinised those changes in the trends using the identification of the mean change point to assess their changes with the construction of the barrages. Lastly, the chapter has found the dominant variables in water quality data in the rivers and understood the distribution of observations as to those variables in ordination space. The key findings are:

1. Meteorologically dry conditions in the Nakdong catchment are more dominant in the 2010s than in the 2000s with two dry years observed in 2015 and 2017. A cumulative precipitation plot shows that a lack of summer precipitation is the main feature observed in dry years. The dry year of 2015 is more severe in the north of the catchment, whilst the dry year of 2017 is in the south of the catchment. The changes in precipitation are well-represented in the response of the hydrological system and the impacts of dry conditions are more severe in the tributaries than in the main river.

2. The annual mean discharge in the main river and the tributaries shows a great response of discharge to the hydrological drought events but does not show a clear reaction to the construction of the barrages due to the timescale. However, the monthly mean discharge shows that the construction of the

barrages has brought a significant change to the hydrological system in the main river with a reduction of the summer peak and the increased winter discharge. These changes are not detected in the tributaries.

3. The time-series trends of water temperature in the rivers are similar to the change of air temperature, but water temperature has regional variations, particularly in the tributaries. On the time-series trends of water temperature, a change point between 2012–2013 is detected at sites located in between the barrages in the main river, but no change point is evident in the tributaries. Through this change point, the water temperature in the main river started to behave in the same way. The higher correlations after this change point at the time when the barrages were completed in April 2012 suggest that the construction of the barrages has made a significant impact on water temperature in the main river through impoundment and an increase in water volume and retention time. Meteorological droughts in 2015 and 2017 would have contributed to the increase in water temperature as well. However, the water temperature in the tributaries has remained independent of the changes.

4. The annual trend of water quality between 2001 and 2018 exhibits different trajectories in the main river and the tributaries. During the whole monitoring period, the main river has parameters like pH, DO, BOD, COD, and Chla starting to increase around 2010–2012, whilst the tributaries remain similar or slightly decrease in those variables. The SS has a different pattern between the rivers. The SS level in the main river significantly decreased in 2013 and remained low after the peak during 2010–2012, however, the tributaries remained similar after a smooth peak during the same period. In contrast, TP concentration in both rivers has seen a significant decrease since the early 2000s. Overall, these results show that water quality in the tributaries changes little or remains similar during the whole period except for TP concentration. However, the main river has a significant change in

variables (pH, DO, BOD, Chla, and SS) between 2010–2012, which appears to be related to the construction of the barrages.

5. Using the identification of mean change points in the trends, the spatio-temporal pattern of water quality in the main river is examined with the impact of the barrages. The results demonstrate that variables like pH, DO, Chla, and COD have started to increase with a change point at the time of the construction or the completion of the barrages. The extent of an upward change is more significant at upstream sites M4–M8 than those in the mid-, and downstream. Also, SS is directly influenced by the construction of the barrages. The SS level started to rise in 2010 and formed a peak during 2010–2011. After the construction was completed in April 2012, the SS level came down and dropped further below the pre-construction level from 2013. In the meantime, TP concentration in the main river has seen a decrease in all sites. This improvement in TP concentration is also identified in the tributaries where no barrages were constructed. As a result, a decrease in TP in the catchment is attributed to other efforts to tackle nutrient enrichment such as the installation of sewage treatment plants and the introduction of a stricter criterion on phosphorus level in released water.

6. The results of Principal Component Analysis (PCA) show that nutrients and organic matter are dominant variables controlling the principal components in water quality data from 2001 to 2018 in the Nakdong catchment. According to their gradients, sites in the main river are aligned in geographical order with sites M10–M12 being the most polluted due to the influx of pollutants from tributary KH. This downward distribution in the main river can be viewed as having continuous gradients of physical variables at the catchment scale or as having downward arrays of patches, controlled by different factors such as hydrogeomorphic processes at the reach scale. In the tributaries, a downward gradient is identified between two sites within each river, but it is not as significant as in the main river due to limited spatial extent.

Understanding these structures in the Nakdong river is vital because they can provide physical habitats and determine the distribution of diatoms in the rivers.

3.7 Summary

This chapter has examined the spatio-temporal pattern of water quality in the Nakdong river and its tributaries over the last 18 years in response to changes in climate and hydrological systems including the construction of the barrages. The examination of water quality in the Nakdong catchment is important as it provides not only the impacts of the barrages on water quality but also provides a basis for diatom analysis, which will be examined in Chapter 5.

Ahead of it, the replicability of diatom assemblages from multiple random rock samples at the reach scale is to be examined in Chapter 4. This analysis will demonstrate variability in diatom assemblages from random sampling and what the variability means in water quality assessment. These results will make the use of diatoms more robust in water quality assessment.

Chapter 4

Replicability of Diatom Assemblages

4.1 Introduction

This chapter aims to investigate the replicability of diatom assemblages among random rock samples at a reach scale in the Nakdong river in an attempt to answer research question 2 (section 2.6.2). In water quality assessment studies using epilithic diatoms, multiple rock samples are widely used as substrata over macrophytes in the targeted-habitat method as they are naturally present in most rivers regardless of season (Round, 1991). During the sampling process, it is common to make a composite sample by merging diatoms collected from multiple rock samples within a reach in the belief that multiple random samples by one habitat type within a reach must have similar diatom assemblages (Kelly et al., 2009; Stevenson et al., 2010).

To date, however, compared to previous research looking at differences in diatom assemblages between habitat types within a reach, variability of diatom assemblages among random rock samples collected at a reach scale has received little attention with only two specific studies (Passy, 2001; Kahlert and Savatijević Rašić, 2015). Although there is a general recommendations for sampling in the field (Kelly et al., 1998), the selection of multiple rock samples in the field relies on the judgement of researchers, which is highly subjective. Therefore, there is a need to evaluate variability of diatom assemblages at a reach scale and potential sources of variability so that the results of this project and existing diatom data can be more meaningful and reliable.

To fulfil the aim, multiple rock samples at the same site within a reach will be randomly collected at the same time in the Nakdong river and their variability in diatom assemblages between and within sites will be statistically examined. Ultimately, the demonstration of the replicability of diatom assemblages in random rock samples will provide a firm ground for diatoms to be accurately employed in assessing river water quality as an environmental indicator.

4.2 Methods and datasets

In order to achieve the aim , methods, dataset, and expected outcomes are presented in the flow chart (Figure 4.1). More detailed methods are explained below.

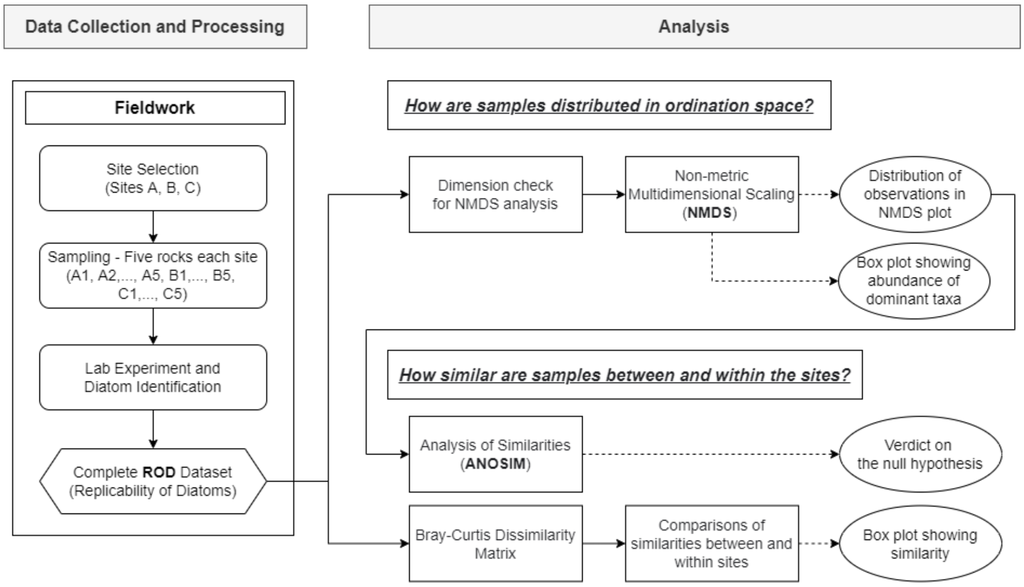


Figure 4.1: Data requirements and workflow to examine the replicability of diatom assemblages

4.2.1 Rock sampling for diatoms

Five cobble size of rock samples with biofilms on the surface were collected at each of three sampling sites A, B, and C in the Nakdong river in December 2018 (Figure 4.2). Three sites were selected to ensure that geographical variations at the broad scale are encompassed with each site from upper-, mid-, and downstream of the Nakdong river and they are also government diatom monitoring sites. In addition, they were selected in consideration of a geographical distance from the barrages and the confluence of the tributaries to minimise possible effect of them on diatoms.

Site A is 20.5 km upstream from the first barrage (B1), and has a free-flowing water in the upstream of the main river, joined by tributary NS (a Strahler stream order 8), while sites B and C are located in between the barrages in the main river.

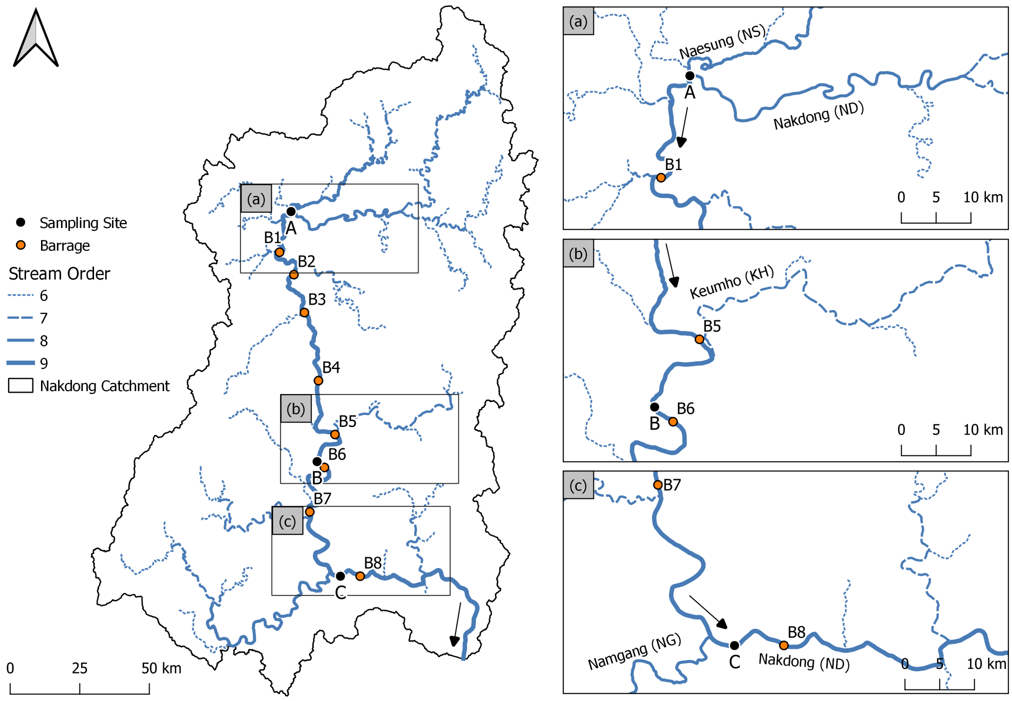


Figure 4.2: Map showing sampling sites for the ROD dataset. (a), (b) and (c) are inset maps at a large scale for sampling sites A, B, and C, respectively. Arrows indicate the flow direction.

Site B, located in between B5 and B6, is 15.3 km down from the junction where tributary KH (a Strahler stream order 7) flows into the main river and site C is 3.8 km down from the junction where tributary NG (a Strahler stream order 8) meets the main river. Site A in the main river has a Strahler stream order of 8, and sites B and C have 9.

In the field, rock sampling at all three sites was carried out at the riverside because the main river is not wadeable. Size and shape of rocks, and their location in relation to any obstacles, for example, bridges or trees that can influence the growth of diatoms by obscuring sunlight, were carefully considered to ensure that sampled rocks were submerged in the water and their upper surface was large and flat enough for diatoms to grow (Round, 1991; Kelly et al., 1998; Lee et al., 2016). Rock samples were taken to a laboratory where their surfaces were rubbed off with a brush and washed off with distilled water before the water was kept in a sampling bottle. The sampled water was then preserved with 10 % formaldehyde before they were to be made into diatom slides.

4.2.2 Diatom slide preparation and identification

The Replicability of Diatoms (ROD) dataset was created through preparation and identification of five diatom samples (1–5) each from three sampling sites A–C and there were 191 different taxa identified in total.

According to the guidance in Korea (Lee et al., 2016), permanent diatom slides were prepared using the sulphuric acid method, which is the same method used for the GOVD dataset by research team in South Korea. All diatom frustules were then identified at species level at 1000x magnification and counted up to a minimum of 300 valves per slide (Bate and Newall, 1998). Diatom reference atlases used were the same as those in the government diatom research (Watanabe et al., 2005; Lee et al., 2016; Joh, 2010; Joh et al., 2010; Joh, 2011; Lee, 2011; Joh, 2012a; Lee, 2012b; Joh, 2012b; Lee, 2012a; Joh, 2013; Joh, 2015a; Joh, 2015b).

4.2.3 Multivariate analyses

The established ROD dataset is a matrix of counts for 191 taxa in column from 15 observations in row. The dataset was converted to relative abundance for multivariate analyses. As the aim of this chapter is to empirically demonstrate the replicability of diatom assemblages in random rock samples, all diatom taxa identified were included regardless of their abundance to ensure that all variability in the ROD dataset is examined.

4.2.3.1 Non-metric Multidimensional Scaling (NMDS) analysis

For analysing community structure in multivariate ecological data, ordination methods which consider communities on the basis of the identity of the component species as well as their relative importance in terms of abundance or biomass have been widely used (Clarke, 1993). Non-metric multidimensional scaling (NMDS) analysis is a technique used to visually display observations based on non-Euclidean distance in a low dimensional space (Oksanen et al., 2019).

First of all, the dimensional space for the ROD dataset in NMDS analysis was determined. This is to ensure that the observations can be projected with the distances among them kept within an acceptable range of stress value (stress value < 0.2 is regarded as usable) (Clarke, 1993). This step was performed using the dimcheckMDS command in the vegan package (Oksanen et al., 2019) with the Bray–Curtis dissimilarity method. On the obtained dimensions, NMDS ordination for the ROD dataset was projected using the metaMDS command to compare the observations in ordination space. The distribution of the 15 samples in ordination space was understood with comparisons of dominant taxa per site in relative abundance. This helped understand variations of dominant species per site in NMDS space with actual relative abundance figures.

4.2.3.2 Analyses of similarity

After using the ordination method, there were two different types of statistical analyses carried out to statistically evaluate similarities of the 15 samples between and within the groups in the ROD dataset. First, Analysis of Similarities (ANOSIM) was performed using the anosim command in the goeveg package (Goral and Schellenberg, 2018) to test for statistical significance in the dissimilarities between and within the groups based on dissimilarity ranks.

Second, Bray–Curtis dissimilarities between the 15 samples were calculated using the vegdist command in the vegan package (Oksanen et al., 2019) to compare their similarities between and within the sites. To make it intuitive to understand the similarities, the obtained dissimilarities were then converted to similarities that were obtained by subtracting the values from 1 before multiplying by 100 to treat them as percentages. For example, dissimilarity between A1 and A2 is 0.261 and their similarity is 73.9 %. The result of the similarities among the 15 samples was visualised by sampling sites as box plots to examine the similarities between and within the groups.

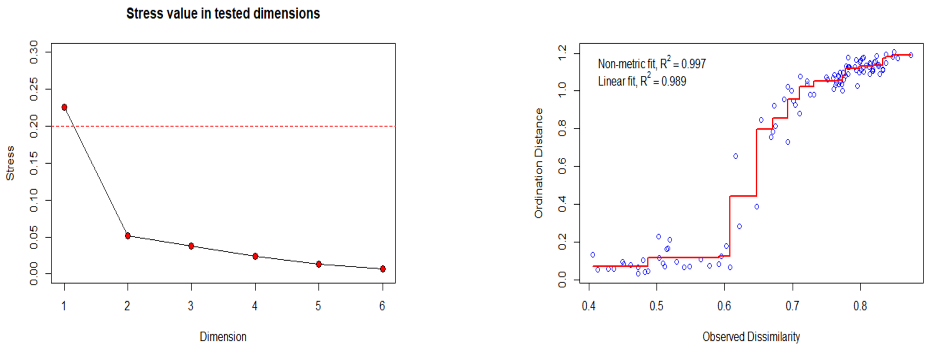
4.3 Results

4.3.1 Distribution of the ROD samples in NMDS space

The analysis of dimension check for the ROD dataset confirms that the observations can be projected in two dimensional NMDS space with an excellent level of stress value (see Figure 4.3 (a), stress value = 0.05). At the selected dimensional space, 99.7 % of the observations can be best projected in non-metric fit with no prospect of misinterpretation (Figure 4.3 (b)).

Figure 4.4 shows the distribution of 15 samples (A1–A5, B1–B5, and C1–C5) and species greater than 1 % in abundance in two dimensional space. The observations are clustered within the groups, but well-dispersed between the groups in space. Particularly, samples in sites A and C are considerably clustered in close proximity within their groups, while site B has a elongated shape of distribution with B2 significantly different from the others. Moreover, distances between each sample in site B are generally greater than those in sites A and C.

Out of 191 taxa in the ROD dataset, 31 taxa that are greater than 1 % in abundance are on display in Figure 4.4 and they account for 73.1 % in abundance in the whole dataset. Each site is variously characterised by different types of diatom species (Figure 4.5). For example, site A is mainly comprised of *Achnanthes minutissima*, *Amphora pediculus*, *Cymbella turgidula*, *Diatoma vulgaris*, *Fragilaria crotonensis*, and *Synedra tenera*, while site C has *Aulacoseira alpigena*, *A. ambigua*, *A. granulata*, *Cyclotella pseudostelligera*, *Cyclostephanos invisitatus*, and *Stephanodiscus hantzschii* in dominance.



(a) Stress value at different dimensions

(b) R^2 comparisons

Figure 4.3: Stress plot and Shepard diagram to determine dimensions for the ROD dataset. Red dashed line in (a) indicates a threshold for the stress value.

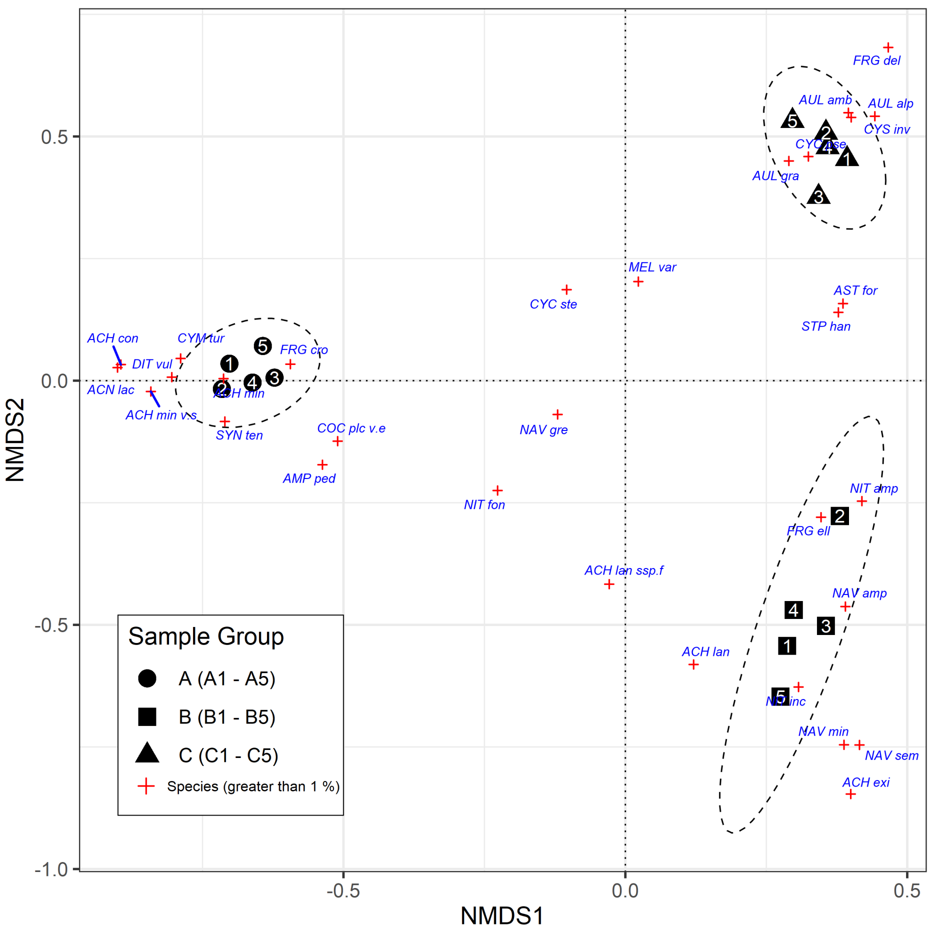


Figure 4.4: Non-metric multidimensional scaling (NMDS) biplot of the ROD dataset. Species greater than 1 % in relative abundance are on display. Dotted lines are arbitrarily drawn for ease of reading and dashed ellipses represent 95 % confidence area.

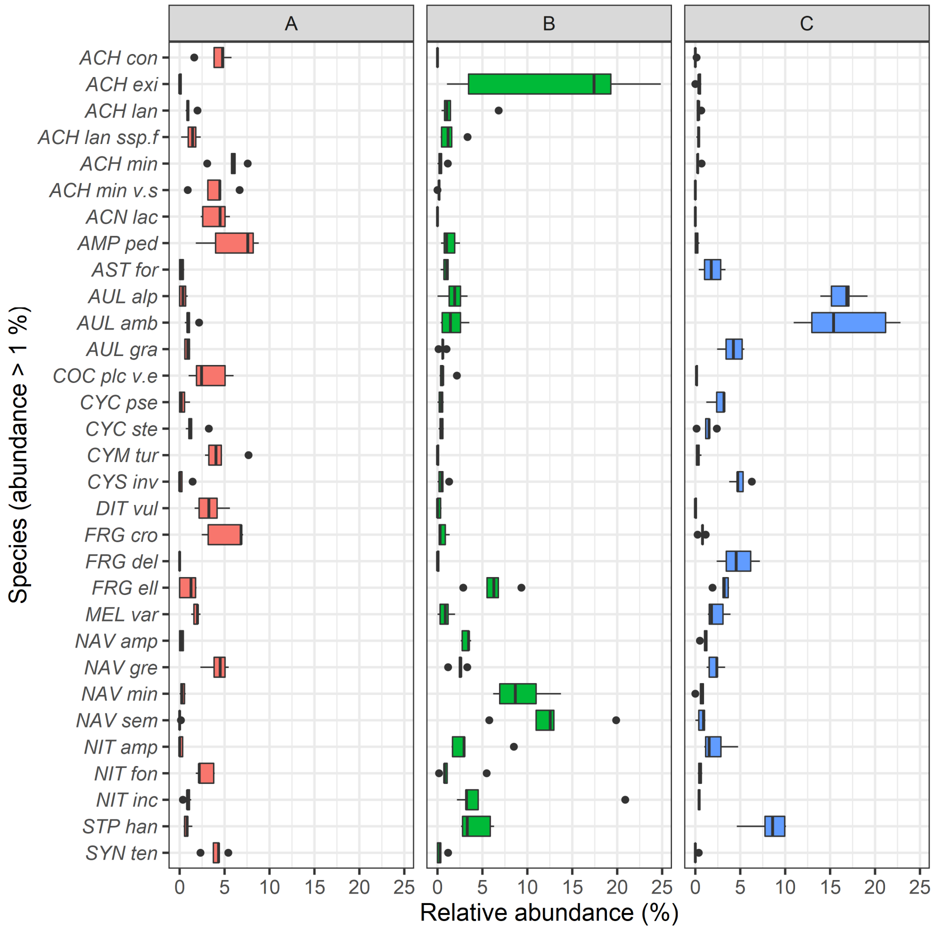


Figure 4.5: Comparisons of dominant diatom species in abundance between sampling sites A–C in the ROD dataset. Species greater than 1 % in relative abundance are on display. The number of observations per group is 5.

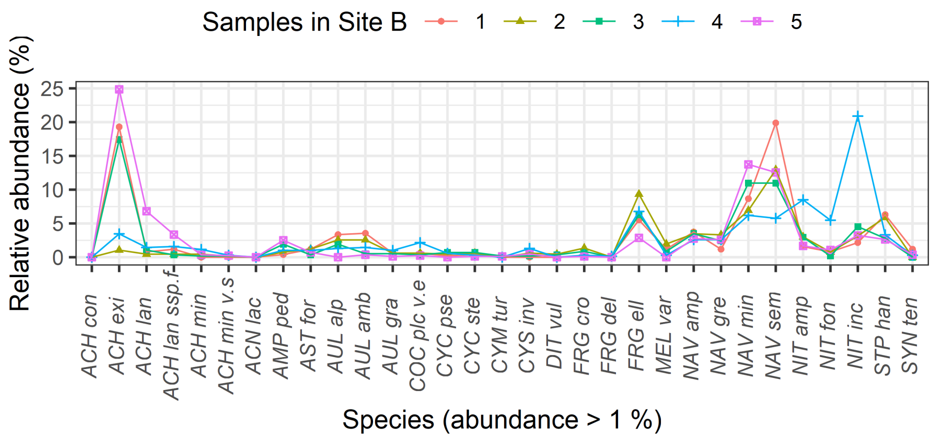


Figure 4.6: Distribution of relative abundance of dominant diatom species in five samples (B1–B5) in site B.

Contrarily, site B has more variations in community structure within the group. Overall, all samples in B have primarily *Achnanthes exigua*, *Fragilaria elliptica*, *Navicula amphiceropsis*, *N. minima*, *N. seminulum*, and *Nitzschia inconspicua* being dominant, while B2 and B4 have somewhat different patterns of abundance with fewer species (Figure 4.6); both samples have significantly low abundance (1.1 % and 3.5 %) of *Achnanthes exigua* compared to other samples in the group, and B2 has a relatively high abundance of *Fragilaria elliptica*. Sample B4 is characterised by a low abundance of *Navicula seminulum* and a high abundance of *Nitzschia amphibia* and *N. inconspicua*.

4.3.2 Similarities between and within the groups

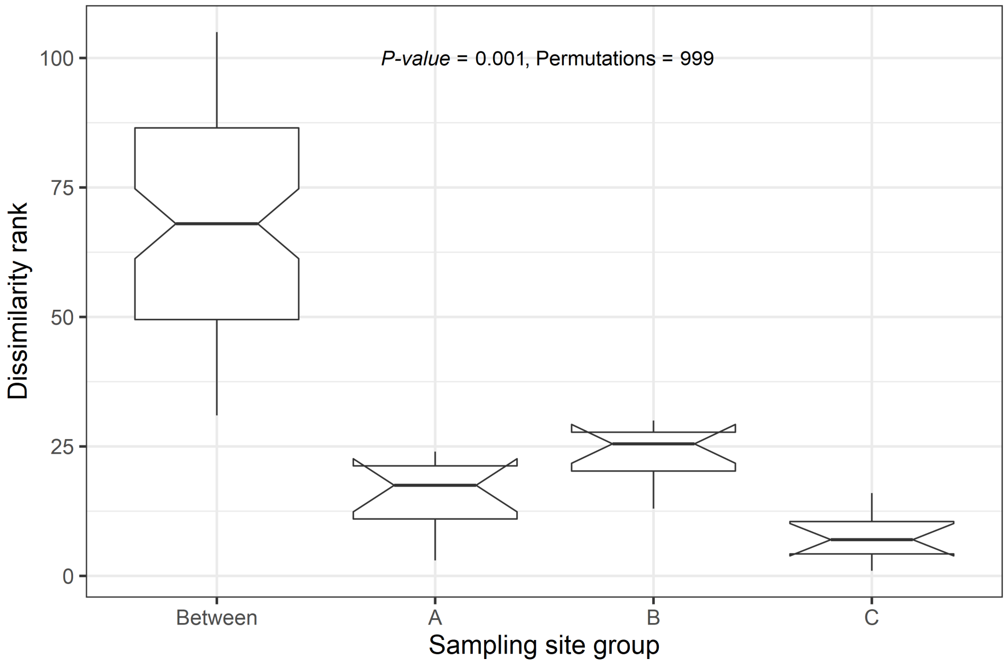


Figure 4.7: A notched box plot showing the results of Analysis of Similarities (ANOSIM) for the ROD dataset based on dissimilarity rank

The distribution of the 15 samples in ordination space by dissimilarity ranks was tested by the ANOSIM. The results of ANOSIM suggest that the null hypothesis that the similarity between the groups is equal to the similarity within the groups should be rejected ($p = 0.001$). Therefore, we can conclude that there is statistically a significant difference in the similarities between the between-groups and the within-groups. The analysis also confirms that ANOSIM statistic R is 1, which means that there are more similarities among samples within the groups than between the groups. Between the groups, the three sites have different values for the medians in dissimilarity ranks, obtained from the Bray–Curtis dissimilarity method, although the upper quartile in site A and the lower quartile in site B overlap marginally (Table 4.1).

Table 4.1: Quartile range of dissimilarity ranks between and within the groups in the ROD dataset

Group	0 %	25 %	50 %	75 %	100 %	N
Between	31	49.50	68.0	86.50	105	75
A	3	11.00	17.5	21.25	24	10
B	13	20.25	25.5	27.75	30	10
C	1	4.25	7.0	10.50	16	10

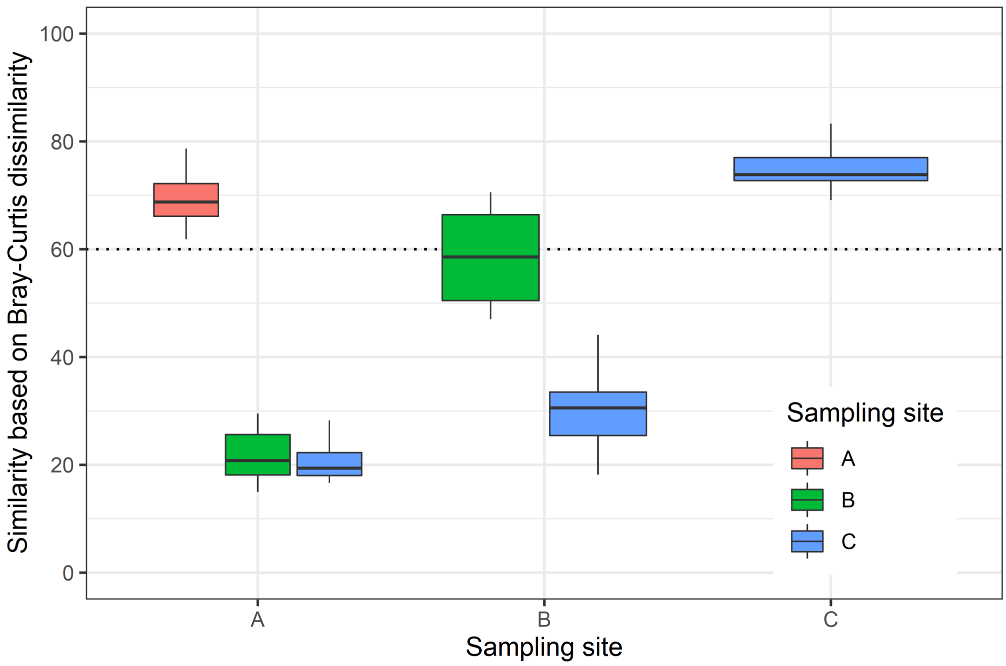


Figure 4.8: Comparisons of similarities between and within the sites for the ROD dataset. Similarity was obtained through subtracting dissimilarity value from 1 before multiplying 100. Dotted line indicates a threshold of 60 % similarity for being considered as replicate samples in quality control of diatoms (Kelly et al., 2009).

Figure 4.8 displays similarities between and within the sites for the 15 samples in the ROD dataset. It shows that samples in site C are the most similar to each other within the groups with a mean of 75.2 % in similarity, followed by 69.3 % for site A, and 58.9 % for site B and the mean similarity between the three sites is 67.8 %. Compared to sites A and C showing little variation, site B has relatively big variations with the lowest group mean. On the other hand, similarities between the groups are low, ranging from 15 % to 45 %.

4.4 Discussion

4.4.1 How similar should diatom assemblages be to be considered as replicate samples?

The experiment with five rock samples randomly taken at sampling sites A–C in the Nakdong river demonstrates that the structures of diatom communities within the groups are statistically more similar than those between the groups. It also demonstrates that epilithic diatoms attached to the surface of rocks can have a range of 58–75 % similarity with a mean of 67.8 % through multiple random sampling at the same site at the same time.

The spatial and temporal variability in diatom assemblages at a reach scale was well-studied by Snell (2014) in headwater streams in the U.K. In her research, diatom assemblages were distinct between the riffle and pools and physical attributes such as depth, velocity, and substrate are the key variable contributing to the difference in diatom assemblages diversity and chlorophyll-a concentration. In particular, in the pool, variability in the TDI was big from "High" to "Poor" trophic status and consideration of the location of rock samples is an important element for accurate assessment. However, this study does not provide any specific figure for a threshold of similarity in diatom assemblages. As a result, it is not possible to conclude that the current range of similarity in this project is sufficient to be considered reproducible. Nevertheless, we may take a hint from previous studies. Wu et al. (2016) adopted 60 % in similarity as a threshold for diatom assemblages to be considered similar in their research. Kelly (2001) examined the similarity of diatom assemblages between the primary analyst and auditor at identification for quality control of diatom samples and suggested that as a rule of thumb, a Bray–Curtis similarity value of 60 % is acceptable for most samples to be accurately assessed in the ecological quality index using the TDI as an example. Kelly (2001) argued that a threshold should be increased up to 70 % for samples with low levels of diversity in assemblages. On a similar rationale, even a stricter threshold of 75 % is applied in the South Korean monitoring programme for quality control of diatom samples (Watershed and Ecology Research Team, 2015).

Judging from these figures, it can be cautiously concluded that sites A and C are above the conventional criterion of 60 % to be considered replicate samples, while site B is below the limit. Thus, samples in site B cannot be considered replicate samples. However, it is difficult to conclude that 60 % similarity is the right threshold for replicate samples because it is little known about what percentage of similarity is naturally expected to be in random sampling. Therefore, there is a need to extend this experiment further with a consideration of more sampling sites taken at adjacent reaches for better statistical significance.

4.4.2 What causes the heterogeneity of diatoms within the groups?

Like the assumption that the application of the targeted-habitat method for sampling can capture the similar structure of diatom communities (Kelly et al., 2009; Stevenson et al., 2010), samples within sites tend to be similar to each other, but there is still a degree of variability in diatom assemblages, particularly in site B. This suggests that there is a possibility that random sampling within a reach by one habitat type can potentially capture big variability in the composition of diatom assemblages.

It is extremely rare to have the identical structure of diatom assemblages from random rock sampling in a river and some degree of variability is almost inevitable because it is a natural experiment in rivers with lots of variables involved in processes. Thus, the heterogeneity of diatoms can arise at a small scale (reach, habitat, substratum, and cellular scales) due to physical, chemical, and biological processes (Stevenson, 1997).

The variability of diatom assemblages in site B could be justified and diatom samples in site B could be used to represent the ecological state of the river at that point after multiple rock samples are merged into a composite sample because diatom samples are collected by the targeted-habitat method within a reach. However, the variability in diatom assemblages within the groups at this site may cause significant errors in assessing the ecological state of the river. Thus, it is important to delve into what might cause variability in site B and what variability means in water quality assessment.

Here, I mainly focus on what can cause the heterogeneity of diatoms within the groups in rivers and what these results imply in applying diatoms in water quality assessments as an example of site B. The heterogeneity of diatoms between the groups will be discussed later in Chapter 6 in relation to the spatial pattern of environmental variables such as climate, hydrology, water quality, land use, etc. at the broad scale (Stevenson, 1997).

The variability of diatom assemblages within a reach may be caused by environmental conditions at the local scale or smaller scales such as microhabitats. Even within a reach of a river, there can be a difference in physical characteristics, which can affect the distribution of diatoms in rivers as reported in the study of Passy (2001). Passy (2001) found that physical variables such as current velocity (38 %) and space (10 %) were more important in explaining the spatial distribution of diatoms than chemical variables at a reach scale ($4\,\mathrm{m} \times 5\,\mathrm{m}$) in a stream where rock samples were collected at intervals of 0.5 m.

At the local scale, there is also a possibility that incoming tributaries may have affected the distribution of diatom assemblages. Confluences in river networks have more diverse morphological features (Benda et al., 2004), which can then provide diverse riverine habitats for diatoms, although no research has been carried out so far in relation to the biodiversity of diatoms at river junctions. Site B is 14 km down from the junction where tributary KH joins and creates sediment bars in the channel of the main river. So, it could be the effect of the tributary, but it seems unlikely because morphological features in the main river have been severely disrupted and homogenised by the construction of the barrages and dredging (Im et al., 2015). Furthermore, water mixing between two rivers at a junction may have been a cause as Lane et al. (2008) demonstrated that water mixing processes for two large rivers in Argentina can be complete as short as within 8 km of the confluence, with rapid mixing or as long as 400 km at slow mixing depending on bed morphology and momentum ratio.

The variability may also arise from unintended errors in the sampling process. Even though precautions were strictly taken as much as possible in the fieldwork according to the recommendations (Kelly et al., 1998; Watershed and Ecology Research Team, 2015), it is not always possible to take samples of rocks that are similar in size and shape, which could potentially offer different microhabitats for diatoms. For example, Kelly et al. (1995) found significant differences in diatom assemblages and indices between samples from cobbles and boulders in a low nutrient site in England. In this project in the Nakdong river, five rock samples in site B were relatively small in size and less flat in shape compared to others in sites A and C and this difference in substrata might play a role in the different composition of diatom assemblages. It is also worth noting that small rocks are more likely to move during river flow. Thus, it is not possible to entirely rule out the possibility that some rock samples at site B are allochthonous.

To sum up, variability in diatom assemblages among random samples within a reach can be triggered by either differences in environmental conditions at the local or smaller scale or unintended errors in the sampling process. To better use diatoms in water quality assessments, it is necessary to experimentally study the variability of diatom assemblages among multiple random samples in the river network of interest.

4.4.3 What does the heterogeneity within the groups imply?

Adopting the criterion of 60 % similarity from the study of Kelly et al. (2009), the mean similarity of 67.8 % in the ROD dataset is sufficient to claim that samples within the sites are considered replicate samples. However, the similarity of 58.9 % among samples in site B is questionable, because it is marginally below the limit. In this case, it is necessary to scrutinise what species make differences and what the difference potentially means in applying diatoms in water quality assessments. Here, I mainly focus on variability in diatom assemblages in site B as an example and discuss what disparities would mean when it comes to assessing water quality in rivers.

The disparities in diatom assemblages are noticeably observed with samples B2 and B4 in comparison to the rest of the samples in site B (Figure 4.6). However, despite some species having a noticeable difference in relative abundance, they may still have similar environmental preferences. For example, samples B2 and B4 have a low abundance of *Achnanthes exigua* compared to others on the site. Instead, B2 has a relatively high abundance of *Fragilaria elliptica* and *Navicula seminulum*, and B4 has a high abundance of *Nitzschia amphibia* and *N. inconspicua*. In the literature, *A. exigua* is known to have a tolerance to nutrient and metal pollution (Pandey et al., 2015; Lee et al., 2016) and its dominance in samples B1, B3, and B5 indicate highly polluted water conditions in site B. Similarly, *N. amphibia* and *N. inconspicua* identified in sample B4 are the typical species frequently found in alkaline water as well as eutrophicated water (Watanabe et al., 2005). The occurrence of *N. seminulum* in sample B2 is also closely related to high pollution in water. Therefore, despite the different types of species in the presence of samples in site B, they are all indicative of a high level of pollution. In addition, it is reported that *A. exigua* and *N. amphibia* have the same sensitivity ($S = 5$) and indicator value ($V = 3$), while *N. seminulum* has 4 and 1 for each in assessing water quality using the modified Trophic Diatom Index (mTDI) in Korea (Watershed and Ecology Research Team, 2015, p.25). Thus, in this case, the difference between the taxa would make little impact on the results of the mTDI due to a trade-off in the process.

Diatoms also can be used at the genus level to assess environmental conditions in rivers using their abilities to tolerate nutrient limitation and physical disturbance, often distinguished by growth form (Wang et al., 2005), life-form (Berthon et al., 2011) or ecological guilds (Passy, 2007b).

The proportion of growth forms — prostrate, erect, stalked, unattached, and motile — in diatom assemblages can be used as a means of water quality assessment as different types of growth forms tend to display contrasting responses to impairment (Wang et al., 2005). According to the classification in the study of Wang et al. (2005), *Achnanthes* spp. and *Navicula* spp. are classified as prostrate type and *Nitzschia* spp. is classified as either prostrate or motile, and these three genera are all expected to increase with impairment. Thus, this means that different distribution of species in samples B2 and B4 would be derived from growth forms, rather than the difference in water quality. Similarly, the division of diatoms by life-forms — mobile, colonial, tube-forming, stalked, and pioneer — can be used for trophic level and organic pollution. Berthon et al. (2011) revealed that colonial, tube-forming, and stalked diatoms showed a significant difference in relation to the trophic level and organic pollution in rivers, while mobile diatoms containing the three genera at B2 and B4, and pioneer diatoms did not. In this case, other environmental variables in rivers rather than water quality variables might contribute to the differences in samples B2 and B4.

Likewise, according to ecological guilds (Passy, 2007b), low-profile and high-profile diatoms tend to thrive when trophic levels and organic pollution are low, while motile diatoms are more abundant in polluted water. With this division applied to samples in site B, the occurrence of *Achnanthes exigua* in the low-profile guild at B1, B3, and B5 contrasts with the appearance of *Navicula seminulum* and *Nitzschia amphibia* belonging to the motile guild at B2 and B4. This is because some species within the same genera may have a contrasting ecological preference (e.g. *Achnanthes exigua* and *Achnanthes minutissima*). Therefore, differences in diatom assemblages need to be assessed carefully and divided on a species-by-species basis to use the traits of diatoms at the genus level.

Meanwhile, variability in diatom assemblages among samples in a site may result in a wrong assessment of water quality if the structure of diatom assemblages is directly compared based on a dissimilarity matrix. For instance, in site B, although *A. exigua* and *N.amphibia* share the same ecological preference for high pollution water, but a comparison of samples dominated by different proportions of the two taxa would make difference in the result of the dissimilarity matrix, resulting in a wrong assessment. Therefore, it is recommended not to take a single sample, but to take multiple random samples in rivers and make a composite sample. For example, the structure of diatom assemblage in site B would be dominated by *A. exigua, N. seminulum* with less influence of *N. inconspicua* in a composite sample.

To summarise, the similarity of 58.9 % for five samples in site B, which is below the criterion, can be regarded as replicate samples if the mTDI is to be used for water quality assessment despite samples B2 and B4 having different proportions in some taxa because those taxa have similar ecological preferences with similar or the same sensitivity to pollution. This result means that the heterogeneity of diatoms within a site appears to be related to other environmental variables rather than water quality. The variability among multiple samples in a site would be expected to be less influential when multiple rock samples are collected and made into a composite sample with a few key taxa in dominance. This research is one of the few attempts to examine the variability in diatom assemblages within a reach and it needs more study.

4.5 Conclusions

This chapter has demonstrated the reproducibility of epilithic diatoms in a natural experiment using cobble-sized rock samples by the targeted-habitat method. The experiment was carried out using the ROD dataset established on fieldwork where five rock samples with biofilms on the surface were randomly collected within a reach at three different sites in the Nakdong river at the same time to examine similarities of diatom assemblages between and within the sites. Key findings are:

1. Observations of diatom assemblages from 15 rock samples in the ROD dataset are visually presented in two-dimensional NDMS space with an excellent level of stress value. The 15 samples are well-clustered by the sites but spatially scattered between the sites in ordination space. Five samples in site B have more variations in diatom assemblages than in sites A and C.

2. The three sampling sites have different types of dominant diatom species. For example, samples in site A tend to have *Achnanthes minutissima*, *Amphora pediculus*, *Cymbella turgidula*, *Diatoma vulgaris*, *Fragilaria crotonensis*, and *Synedra tenera* in dominance, while site C is primarily dominated by *Aulacoseira alpigena*, *A. ambigua*, *A. granulata*, *Cyclotella pseudostelligera*, *Cyclostephanos invisitatus*, and *Stephanodiscus hantzschii.* Contrarily, samples in site B have more variations within the group with *Achnanthes exigua*, *Fragilaria elliptica*, *Navicula amphiceropsis*, *N. minima*, *N. seminulum*, and *Nitzschia inconspicua* largely being dominant. In particular, samples B2 and B4 have different patterns of abundance with a few species compared to others; both have a significantly low abundance of *Achnanthes exigua.* Instead, B2 has a relatively high abundance of *Fragilaria elliptica*, while B4 is characterised by a low abundance of *Navicula seminulum* and a high abundance of *Nitzschia amphibia* and *N. inconspicua.*

3. The results of ANOSIM statistically demonstrate that the null hypothesis that similarities between the sites are equal to those within the sites should be rejected. They also confirm that samples within the sites are more similar than those between the sites.

4. Similarities within the sites in the ROD dataset range from 58.9 to 75.2 % with a mean of 67.8 % between the sites. Samples in sites A and C can be considered replicate samples, judging from the criterion of 60 % that is used for quality control of diatom between primary analyst and auditor at diatom identification (Kelly et al., 2009). However, site B is below the limit with a similarity of 58.9 %. Thus, it is important to understand what causes variability and what the variability means in water quality assessment.

5. Despite a targeted-habitat method (rock samples) being applied within a reach for sampling as best as possible, the heterogeneity of diatom assemblages can still arise because of the difference in physical, chemical, and biological characteristics at the local scale or unintended errors in the sampling process. For example, site B is located 14 km down from the confluence where tributary KH joins. As a result, the tributary may have affected diatoms in site B. In the field, rock samples collected in site B are relatively small and less flat compared to those in sites A and C. Hence, sampled rocks may provide different microhabitats for diatoms. Also, there is a possibility that these rocks are allochthonous.

6. Judging by the five rock samples in site B as an example, variability in diatom assemblages among multiple rock samples may make little impact on the result of the mTDI as taxa appearing with different proportions in samples in site B have the same or similar ecological preference in water. Thus, this variability would derive from different environmental factors than water quality. However, the heterogeneity in diatom assemblages may cause a problem when diatom assemblages are to be compared using a dissimilarity matrix.

Therefore, it is recommended not to take a single sample for water quality assessment. Instead, it is recommended to take multiple samples and make a composite sample to minimise the variability in diatom assemblages.

7. Overall, this chapter demonstrates that diatom assemblages from multiple random rock samples within a reach are reproducible at 67.8 % similarity on average, which is beyond the threshold of 60 %. However, it is difficult to estimate what level of similarity is naturally expected to be observed on average in random sampling within a reach because of the limited number of sites in this experiment. It is desirable to expand this experiment further to obtain a more accurate threshold of similarity at a reach scale.

Chapter 5

Diatom Analysis

5.1 Introduction

This chapter addresses research questions 3–5 (section 2.6.2) by dealing with the Government Diatoms (GOVD) and MY diatoms (MYD) datasets. In this chapter, there are three key tasks to be fulfilled. The first one is to assess the performance of the mTDI (modified Trophic Diatom Index) and three diversity indices (Species Richness, Species Evenness, and Shannon Diversity Index) with the GOVD dataset regarding the construction of the barrages. These analyses will answer research question 3 by providing strengths and weaknesses of the current use of diatoms and diversity indices with diatoms in understanding river health in the Nakdong river. Next, the variability of diatoms to the environmental variable within the GOVD sampling period will be examined to answer research question 4 using the MYD dataset that is made up of two samples per site collected three weeks apart on fieldwork. This analysis aims to demonstrate what risks exist within the current diatom sampling period for the GOVD dataset and to highlight what precautions and measures should be taken to observe the current practice. It will also offer insights into how diatoms can be better used in water quality assessment moving forward. Finally, the responses of diatom assemblages in the GOVD dataset will be constructed in time and space and analysed using geometric vectors to answer research question 5. Through these analyses, the responses of diatom assemblages will be understood in relation to various environmental changes including the construction of the barrages in the rivers.

5.2 Methods and datasets

To achieve the tasks above, methods, datasets, and expected outcomes are presented in the flow chart (Figure 5.1). More detailed methods are explained below.

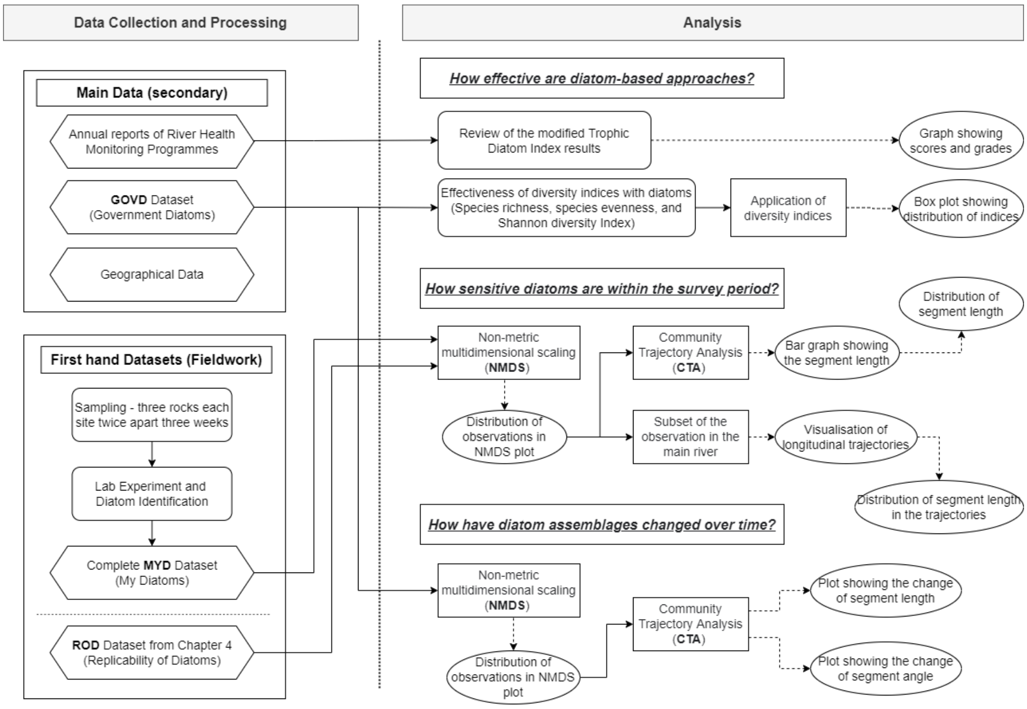


Figure 5.1: Data requirements and workflow for diatom analysis

5.2.1 Evaluation of diatom-based indices

Since 2009, the mTDI has been used twice a year by environmental engineers in South Korea for water quality assessment in rivers. Various studies (Kim et al., 2017; Choi et al., 2019; Lee et al., 2021) report a good performance of the index for the water quality assessment in Korean rivers with a high correlation with pollution in rivers, but their research is primarily focused on the assessment in a short time scale (see Table A.1). In contrast, this project takes a long-term approach to review the results of the mTDI during the whole period in relation to the change in water quality, the environmental changes such as the construction of the barrages, and the hydrological droughts that are demonstrated in Chapter 3. In addition, three diversity indices (Species Richness, Species Evenness, and Shannon Diversity Index) were experimentally applied to the GOVD dataset to test their effectiveness as a tools for river health assessment in connection with the variables above. These analyses provide the advantages and disadvantages of using diatom-based indices as a water quality assessment method when it comes to understanding the changes

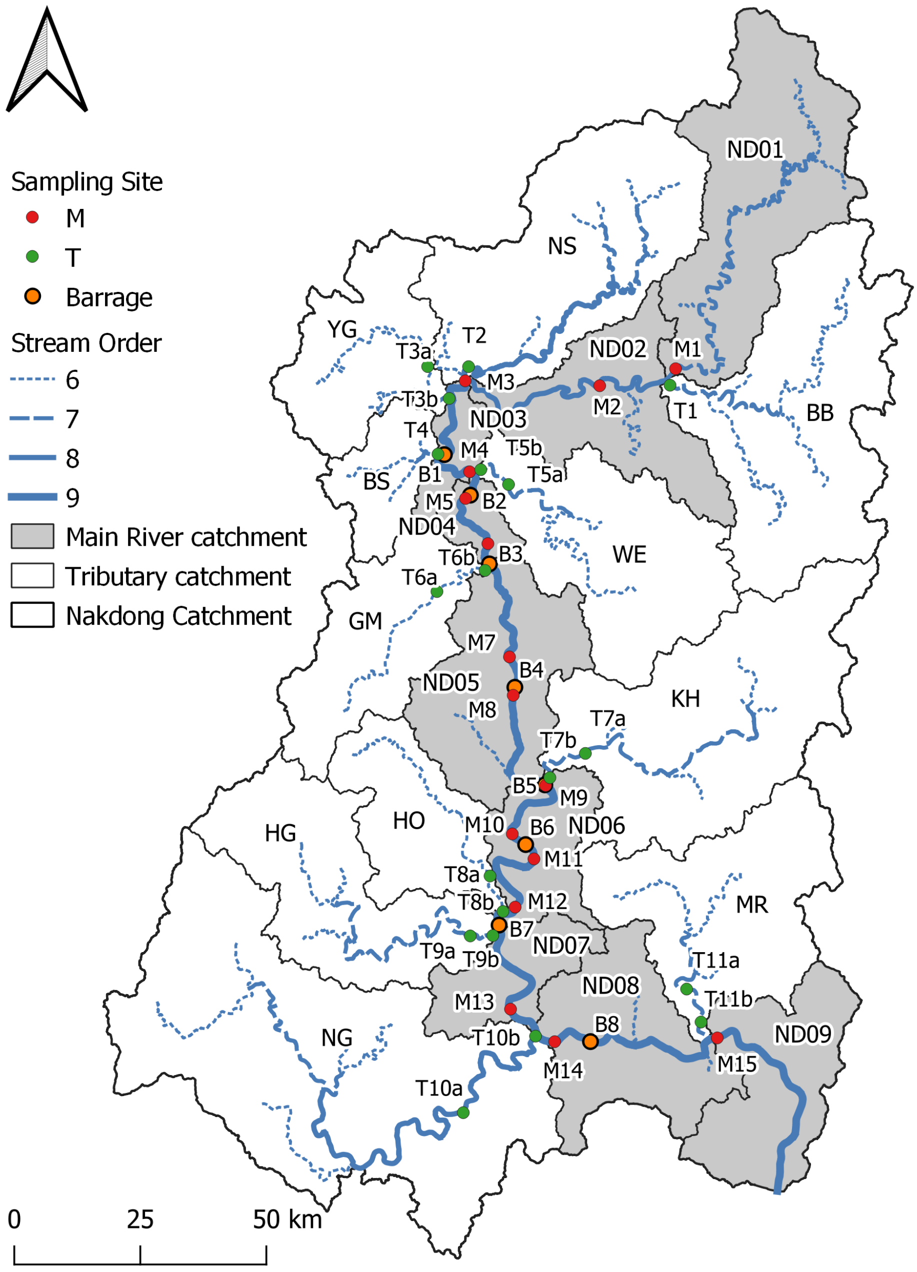


Figure 5.2: Map showing diatom sampling sites for the GOVD and MYD datasets in the Nakdong catchment. Sampling sites M and T indicate the main river and the tributaries. For the MYD dataset, sampling sites M7, M11, T8b, T9a, and T10b were excluded for samplings due to the absence of appropriate rocks or unavailability to access during the fieldwork.

in water quality and riverine environments.

First of all, the mTDI scores from 2009 to 2018 in the study area were obtained from the annual reports of the river health monitoring programme (Lee and Hwang, 2008; Lee, 2009; Hwang, 2010; Hwang, 2011; Hwang, 2012; Hwang, 2013; Hwang, 2014; Hwang, 2015; Hwang, 2016; Hwang, 2017; Hwang, 2018). They were all downloaded from the Water Environment Information System (2019) and were illustrated in plots with their environmental grades. The plots were arranged by catchment (nine for the main river, and 11 for the tributaries: Figure 5.2) because the system only provides averaged scores by catchment by default.

Diversity indices that were applied to the GOVD dataset here are Species Richness (S), Species Evenness (J), and Shannon Diversity Index (H'). The first two are simple and popular among ecologists to describe community structure and assess diversity using the number of the species and the distribution of abundance across the species. Shannon Diversity Index is the most commonly used index in ecology by considering Species Richness and Evenness at the same time (Colwell et al., 2009). The three indices were all applied to the GOVD dataset using functions in the vegan package (Oksanen et al., 2019). The equation for each index is presented below and the results are visualised by stream order and survey over time.

Species Richness (S) equals the number of species recorded at a site and it was simply obtained from running the specnumber command in the package with the dataset. The more species there are, the higher the richness is.

Shannon Diversity Index (H') accounts for both abundance and evenness of the species present in a community and was obtained using the diversity function (Shannon method) in the vegan package through equation 5.1.

$$H' = -\sum_{i=1}^{S} p_{\mathrm{i}} \ln p_{\mathrm{i}} \tag{5.1}$$

where pi is the proportion of species i, and S is the number of species.

Finally, Species Evenness (Pielou's evenness, J) is to assess the distribution of abundance across the species in a community and was calculated using the obtained results of Species Richness (S) and Shannon Diversity Index (H') according to equation 5.2. The value can range from 0 (no evenness) to 1 (complete evenness).

$$J = H'/log(S) \tag{5.2}$$

where S is the number of species.

5.2.2 Variability of diatoms by sampling timing

The sampling method for the current application of the mTDI in water quality assessment in South Korea is solely based upon one-time sampling per site during both the spring (May–June) and autumn periods (September–October) every year (Watershed and Ecology Research Team, 2015). The rationale for this practice is that those periods usually have low flow in discharge before and after the monsoon season in summer following the recommendations (Kelly et al., 1998).

However, variability in diatom assemblages within the periods may still exist in samples depending on the timing of sampling as one-time sampling per site may still capture a different structure of diatom assemblages. Therefore, it is necessary to examine the variability of diatoms within the survey period by comparing multiple sets of diatom samples collected from multiple visits per site. This examination will help understand what environmental variable affects variability in diatom assemblages most between samples collected at different times within the period and what variability means in applying diatoms in water quality assessment moving forward.

Accordingly, two-time samplings for epilithic diatoms were carried out in the Nakdong river and its tributaries three weeks apart during a period from mid-October to early November in 2018 to establish the MY Diatoms (MYD) dataset. The number of sampling sites for the MYD dataset is 29 (13 sites in the main river, and

16 sites in the tributaries), that are the same as those in the GOVD dataset except for five sites (M7, M11, T8b, T9a and T10b) due to unavailability of appropriate rocks or access to the sites during the fieldwork (see the locations in Figure 5.2).

During the fieldwork, three rock samples were randomly collected within a reach at 29 sites. The retrieved rock samples at each site were all made into a composite sample for diatom slide preparations. All other sampling methods and diatom slide preparations followed the same methods used for the diatom analysis in Chapter 4. At the same time, environmental variables for water, such as ORP (Oxidation-reduction potential, mV), TDS (Total dissolved solids, mg/L), DO (Dissolved Oxygen, mg/L), pH, conductivity ($\mu S\,cm^{-1}$), WT (Water Temperature, ℃), resistivity (Ohms) were measured in situ using a multi-parameter probe (YSI 6050000).

The established MYD dataset contains 226 different diatom taxa across 29 sites in total in the catchment and was transformed to relative abundance. As the purpose of this analysis is to assess variability among two sets of diatom samples at each site, all diatom taxa identified at each sample were included without removing rare taxa. The transformed MYD dataset was then merged with the ROD dataset used in Chapter 4 to make comparisons with the ROD dataset as a reference for the magnitude of variability.

First of all, for the comparisons of variability in two-time observations in the MYD dataset and observations collected at the same time in the ROD dataset, a dimension check and NMDS analysis were applied the same as in Chapter 4. The results of NMDS analysis were plotted in NMDS space to make visual comparisons of observations from the MYD and ROD datasets. Afterwards, the results of NMDS analysis were utilised for Community Trajectory Analysis (CTA). CTA is a statistical framework to show spatio-temporal changes of a community based on the geometric analysis and comparisons of community trajectory over time (De Cáceres et al., 2019). Here, two types of length in CTA were measured to examine variability in the MYD dataset using the trajectorylengths command in the vegclust package (De Cáceres et al., 2010). First, segment length, defined as a Euclidean distance between the two observations per site from the MYD dataset in NMDS space, was compared along with distance among observations from the ROD dataset in the same NMDS space. The distribution of segment length in the MYD dataset was statistically examined by river type and stream order in comparison with the result of the ROD dataset with a map presenting the distribution from the MYD data in space. These analyses show the distribution of variability in the MYD dataset compared to the variability observed in the ROD samples taken at the same time. The second type of length is measured among samples in the main river. The result of NMDS analysis for the observations in the main river from the MYD dataset was subsetted and made into two longitudinal trajectories by survey (1st and 2nd) from upstream to downstream sites. The two longitudinal trajectories were compared in terms of trajectory shape and length before the trajectories were broken down to segment length. Segment length, defined as the distance between adjacent sites downstream in the main river, was obtained and its distribution between the two surveys was compared. Then, their distributions were statistically tested using the Empirical Cumulative Distribution Function (ECDF) and the Kolmogorov–Smirnov (K–S) test. These analyses show the distribution of variability between the two surveys in the main river from a longitudinal perspective and demonstrate what variability in NMDS space means statistically.

Afterwards, to find out what variable contributes to such variability most in the MYD dataset during the three weeks, Canonical Correspondence Analysis (CCA) was performed using the cca function in the vegan package (Oksanen et al., 2019) using the MYD dataset paired with the measured environmental variables on fieldwork. This analysis not only reveals what variable causes variability within the survey period but also demonstrates what risks the current sampling practice involves. These results will provide a fundamental basis to conceive a better way to utilise diatoms in water-quality assessment moving forward.

5.2.3 Creation of trajectories for observations in the GOVD dataset

For comparisons of community structures in multivariate ecological data, the identification of the component species and their relative abundance are considered to determine their (dis)similarity (Clarke, 1993). In order to examine changes in diatom assemblages in the GOVD dataset during 2009–2018, ten years of all observations in the catchment were compared based on the Bray–Curtis dissimilarity method. Based on the matrix, Non-metric Multidimensional Scaling (NMDS) analysis and Community Trajectory Analysis (CTA) were carried out using the vegan and vegclust packages (Oksanen et al., 2019; De Cáceres et al., 2010).

The GOVD dataset, downloaded from the Water Environment Information System (2019), consists of observations of diatom assemblages across 34 sites in the study area for a period of ten years from 2009 to 2018 with each year having two-time surveys carried out in spring (May–June) and autumn (September–October). Locations for diatom sampling sites for the GOVD dataset are presented in Figure 5.2. The dataset contains 399 different diatom taxa at species level in density format (cell/cm^2) including four lumped species identified at genus level (*Achnanthes* spp., *Cymbella* spp., *Fragilaria* spp., and *Navicula* spp.). Before the analysis, the dataset was transformed to relative abundance and reduced to 77 taxa by removing rare taxa (smaller than 0.1 % in abundance) and the four lumped species. The selected 77 taxa account for 93.7 % of abundance.

At first, dimensional space for the GOVD dataset was determined using the command dimcheckMDS with the Bray–Curtis dissimilarity method in the goeveg package (Goral and Schellenberg, 2018). This is an important step to project multidimensional observations into reduced dimensional space without distortion of distance in space. In the obtained space, observations in the GOVD dataset were projected using the metaMDS function (Oksanen et al., 2019) to visually compare their locations in NMDS space. The results of NDMS analysis were plotted and compared by river types and sites to see spatio-temporal changes in diatom assemblages in ordination space.

The observations in NMDS space were connected in a longitudinal order (from upstream to downstream) and a time-series order (i.e. stages 1–20 from 2009 spring to 2018 autumn) by river types and sites to form trajectories. Using the obtained trajectories, Community Trajectory Analysis (CTA) was carried out to quantify the extent of changes in diatom assemblages between observations based on the geometric analysis. Here, segment length and segment angle between observations were calculated each based on time-series order as well as survey-based order using the commands trajectorylengths and trajectoryangles in the package (De Cáceres et al., 2010) and compared by stream order. The definitions of types of segment length and angle by the time-series and the survey-based and the equations to calculate them are presented before the results are revealed (see Table 5.1). These analyses enable me to facilitate the comparisons of the changes by various variables using the vectors and to statistically test their significance in an attempt to understand the responses of diatom assemblages to the changes in water quality as well as riverine environments such as the construction of the barrages.

5.3 Results

5.3.1 Evaluation of effectiveness of diatom-based indices

The current use of diatoms for water quality assessment in South Korea was evaluated by looking at the results of the mTDI from 2009 to 2018 in the catchment. The review of the results shows how well the index is performing in terms of explaining the spatio-temporal change in water quality in the rivers as well as the environmental changes in the river network such as the construction of the barrages and hydrological droughts that are already demonstrated in Chapter 3. In addition, the effectiveness of diatom-based indices was tested to explain those changes in the rivers by applying the three most common diversity indices Species Richness (S), Species Evenness (J), and Shannon Diversity Index (H') to the GOVD dataset.

5.3.1.1 Examination of the results of the modified Trophic Diatom Index (mTDI)

Temporal changes in the mTDI scores and their environmental grades from 2009 to 2018 at the catchment level are presented in Figure 5.3 and 5.4 for the main river and the tributaries. They show that the magnitude of changes in the scores and the grades greatly differ by catchment during these ten years.

All catchments in the main river have upward trends in the score albeit with variations (Figure 5.3). For example, two upstream catchments ND01 and ND02 and three downstream catchments ND07–ND09 have changed relatively little with a steady increase in the score, while catchments ND03–ND06 in between them have experienced a remarkable increase over time. Particularly, the latter group has two different patterns; ND03–04 saw a gradual escalation in the score, while ND05–06 saw an abrupt change in 2016.

Those increases in the mTDI scores directly indicate the improvement of river health in the catchment of the main river. According to the environmental grades, the health of the main river has significantly improved over the ten years from a bad or very bad state to a satisfactory level or beyond. Apart from ND01, all other catchments in the main river were below a satisfactory level in the grades with ND03–06 in a very bad state before 2012. Afterwards, ND03–07 saw gradual improvements and reached a satisfactory or good level. However, two downstream catchments ND08–09 have made little improvement in river health, just reaching a satisfactory level over the upper limit of a bad state.

The tributaries have a similar trend in the mTDI scores and their grades as the main river (Figure 5.4). During the same period, none of the 11 catchments has a downward direction in the mTDI scores, but all catchments have improved con-

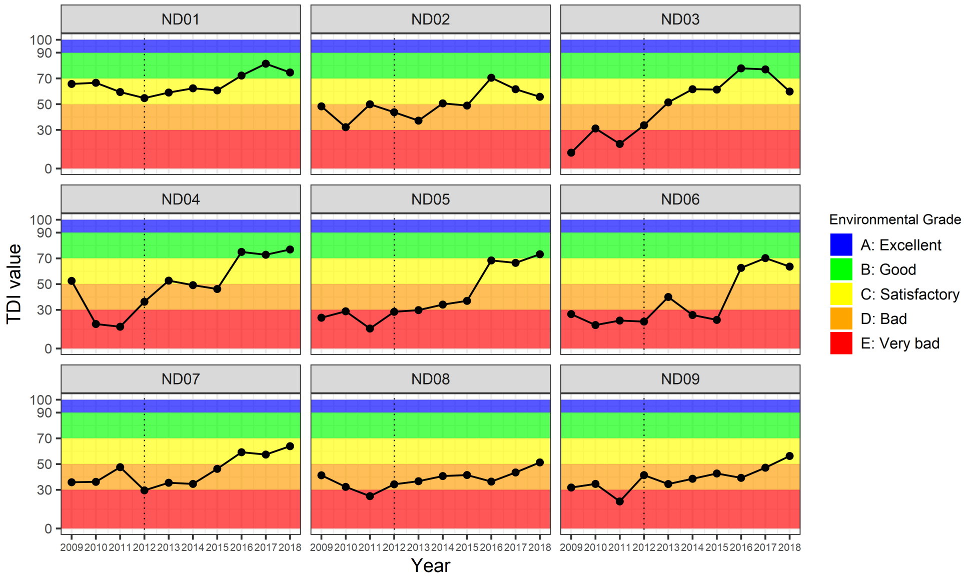


Figure 5.3: Temporal changes in the mTDI scores and their environmental grades from 2009 to 2018 in the main river by catchment. Plots are aligned from the upstream catchment (ND01) to the downstream catchment (ND09) and their locations are presented in Figure 5.2. Vertical dotted lines in 2012 indicate the year when the construction of the barrages was completed. This figure was created using data from the Water Environment Information System (2019).

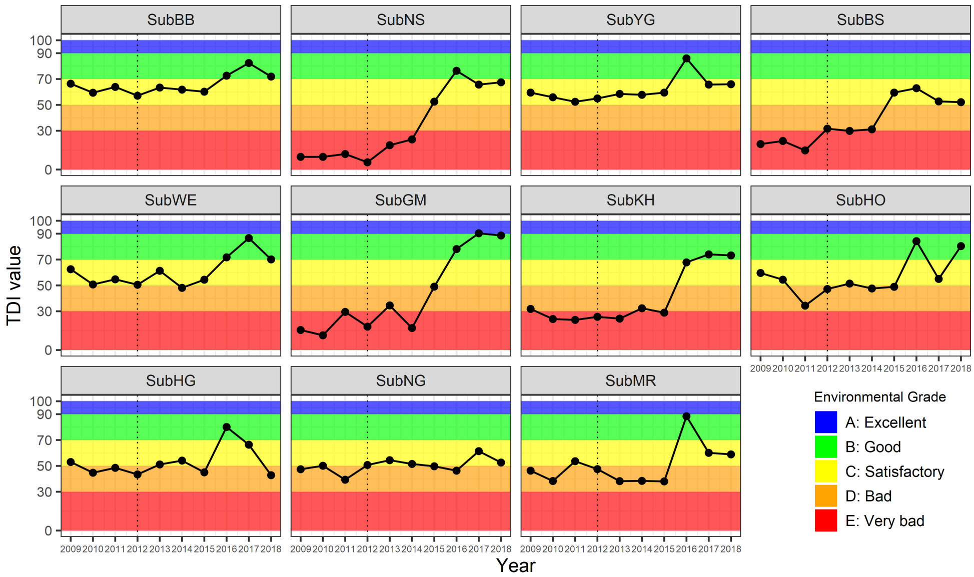


Figure 5.4: Temporal changes in the mTDI scores and their environmental grades from 2009 to 2018 in the tributaries by catchment. Plots are aligned from the upstream catchment (BB) to the downstream catchment (MR) and their locations are presented in Figure 5.2. Vertical dotted lines in 2012 indicate the year when the construction of the barrages was completed. This figure was created using data from the Water Environment Information System (2019).

stantly or abruptly or remained similar. In particular, catchments NS, GM, and KH have an incredibly rapid rise from 2015 or 2016. Other catchments also have an increase in 2016 in their stable trends (e.g. BB, YG, HG, and NG) or their steady upward trends (e.g. WE, BS, HO, and MR).

Similar to the results of the main river, the environmental grades in the tributaries have improved or remained similar. Catchments BB, YG, and WE have remained above a satisfactory level at least after dropping down from a good level in 2017. Catchments NS, BS, GM, and KH were in a very bad state until 2014 when river health is gradually upgraded to above a satisfactory level. In the meantime, catchments HO, HG, NG, and MR from the south of the study area remained between a satisfactory and a bad state in the grade before they saw some improvements between 2016–2017.

In a summary, the spatio-temporal change of the environmental grades in the main river and the tributaries from 2009 to 2018 is presented in a matrix form (Figure 5.5). Overall, river health in all catchments in the study area has been significantly improved to above a satisfactory level over the ten years. The spatial pattern of river health varies by geographical region. Most upstream catchment ND01 and catchment BB have maintained above a satisfactory grade for the whole monitoring period. Other catchments YG and WE in the tributaries follow a similar trend. In contrast, catchments ND02–06 in the main river and catchments NS, BS, GM, and KH are dominated by bad or very bad states of river health until 2016 when the grades have dramatically been improved to a satisfactory or good level. These catchments are located in the mid-region of the study area. During the same time, catchments in the south of the study area have made little improvement in the grade and remained at a bad or satisfactory level.

In these temporal patterns, a simultaneous improvement in the grade in 2016 is noticeable across almost all catchments. In particular, some catchments such as ND02, ND04, ND06, SubGM, SubKH, SubHO, and SubHG were all abruptly upgraded by two grades at once and a very bad state in the grade disappeared in the

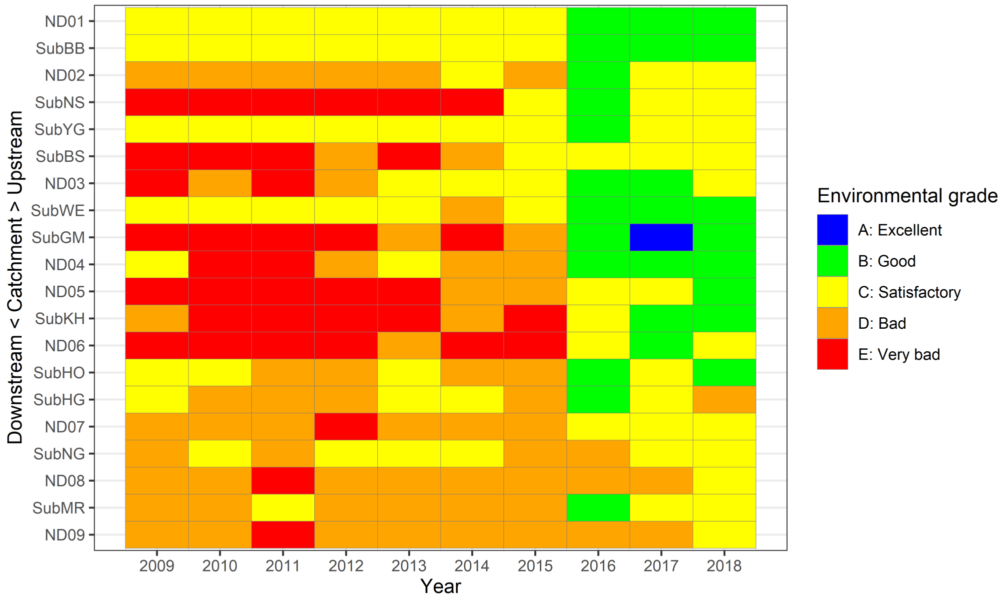


Figure 5.5: Spatio-temporal change in the environmental grades from 2009 to 2018 at catchment level based on the mTDI scores in the Nakdong catchment. This figure was created using data from the Water Environment Information System (2019).

study area. Furthermore, there is no clear signal detected in connection with the construction of the barrages in the main river. Although ND03–05 and ND08–09 recorded a drop to a very bad state in the grade in 2011 that falls within the construction period, however, the impact of the huge scale of the construction is not clearly identified in the environmental grades based on the mTDI results. In addition, there is no strong indication that may be related to the hydrological droughts that occurred in 2015 and 2017 in the Nakdong catchment.

5.3.1.2 The effectiveness of diversity indices using diatoms

Species Richness, Species Evenness, and Shannon Diversity Index are some of the most commonly used-diversity indices in ecology and were experimentally applied to the GOVD dataset. This analysis demonstrates the effectiveness of the indices in explaining changes in water quality in the rivers as well as environmental changes such as the construction of the barrages and the hydrological drought. The analysis began with Species Richness as it is the simplest index based on the number of taxa identified at a site, followed by Species Evenness and Shannon Diversity Index.

Species Richness Species Richness (S) is the simplest index that can be used to determine the diversity in a community based on the number of taxa identified at a site. The distribution of Species Richness in the rivers from 2009 to 2018 is presented by stream order and survey (Figure 5.6: see Figure D.1 for site-by-site details).

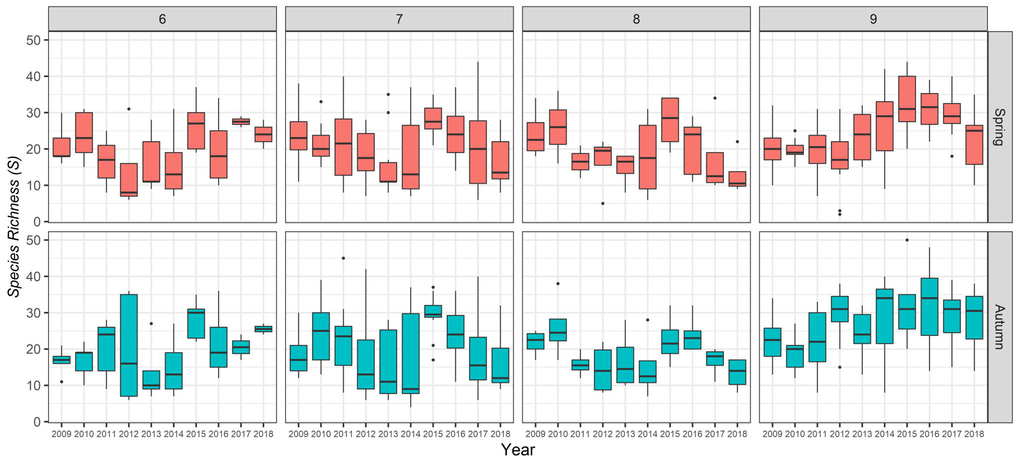


Figure 5.6: Distribution of Species Richness (S) in the GOVD dataset from 2009 to 2018 by stream order and survey

Figure 5.6 shows that the distribution of Species Richness has a similar pattern throughout the time between spring and autumn at all sites. This means that there is no significant difference in the number of species found by survey in spring and autumn. Figure 5.6 also presents that the number of species identified is higher in sites with stream order 9 than in low stream ordered sites in both surveys, however, the difference in the number among sites with stream orders 6–8 is not significant.

The temporal pattern of Species Richness differs by stream order. Sites with stream orders 6–8 have a period in common that recorded relatively a low number of species during 2012–2014 in both surveys. Afterwards, Species Richness dramatically increased up to the peak in 2015 before it came down again. In contrast, sites with stream order 9 display a contrasting pattern by increasing in Species Richness during 2012–2014. Later, they similarly had a peak between 2015–2016 and remained high in the number of species. Judging from these patterns, the number of species identified in sites with stream order 9 remained similar or increased during the construction of the barrages in the main river before it increased in the post-construction period.

Another feature observed in the distribution is that all sites in both surveys have a peak in 2015 when the hydrological drought hit the catchment. These patterns particularly stand out in low stream ordered sites. For example, sites with stream orders 6–8 have a big surge in the number of species in 2015, while sites with stream order 9 also see a similar increase in 2015, but the increase does not look obvious as in other sites.

Species Evenness Unlike Species Richness, diversity in communities can be also assessed as the distribution of abundance across the species identified at a site. Species Evenness (J) ranges from 0 to 1 depending on their distribution of abundance in a community from no evenness to complete evenness across the species. Changes in Species Evenness can be indicative of the emergence or disappearance of species. The distribution of Species Evenness in the rivers from 2009 to 2018 is presented by stream order and survey (Figure 5.7: see Figure D.2 for site-by-site details).

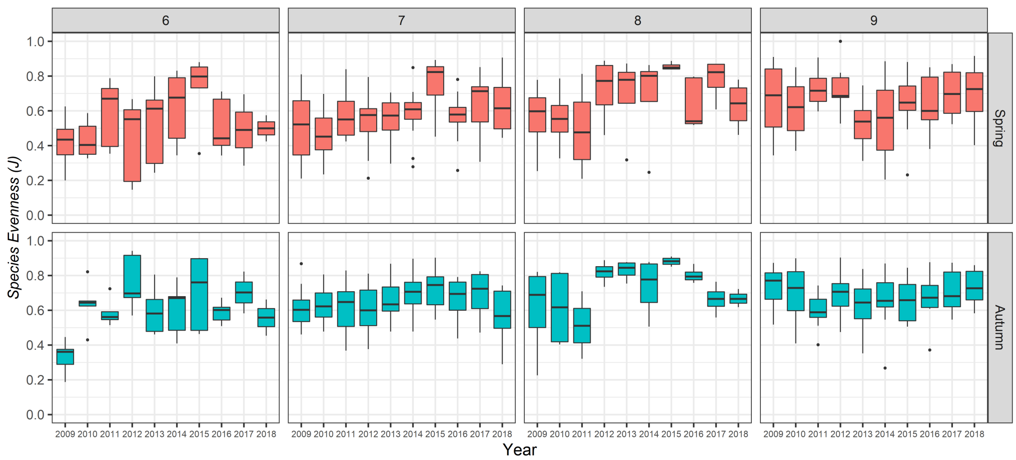


Figure 5.7: Distribution of Species Evenness (J) in the GOVD dataset from 2009 to 2018 by stream order and survey

Figure 5.7 shows that the results of Species Evenness between surveys are fairly similar throughout the time in a range of 0.3–0.8 in both surveys. It also shows that communities in sites with high stream orders tend to have more even distribution across the species than in sites with low stream orders. For instance, sites with stream order 9 have more stable evenness over time at around 0.6–0.8, whilst low stream ordered sites have a bigger variation, ranging from 0.3 to 0.8.

In the distribution of Species Evenness, sites with stream order 9 have a sudden drop in 2013 spring and 2011 autumn, respectively and then gradually increase to the average level. However, during the same time, low stream ordered sites have a different movement in the direction. Sites with stream orders 6 and 8 are characterised by an upward movement in Species Evenness during 2012–2014 with a peak in 2015. Sites with stream order 7 show a similar tendency. However, the peak in 2015 is not identified in sites with stream order 9.

The responses of Species Evenness during the construction period between 2010–2011 are not consistent in sites with stream order 9 between the spring survey indicating the rise and the autumn survey indicating the decline. Following the completion of the barrages in April 2012, the big drop in 2013 spring could be related to the effect of the barrages, but this drop is not obvious in either 2012 autumn or 2013 autumn. In the meantime, the emergence of the peak in 2015 is coincidental with the hydrological drought in the catchment. This peak is well identified in sites with stream orders 6–8, which is similar to the response of Species Richness to the drought.

Shannon Diversity Index Unlike Species Richness or Species Evenness which considers one aspect of community structure, Shannon Diversity Index (H'), considering the number of species and the distribution of abundance across the species together, can be an alternative to assess the diversity in a community. Distribution of the index from 2009 to 2018 is presented by stream order and survey (Figure 5.8: see Figure D.3 for site-by-site details).

Figure 5.8 shows that the distribution of Shannon Diversity Index across the site

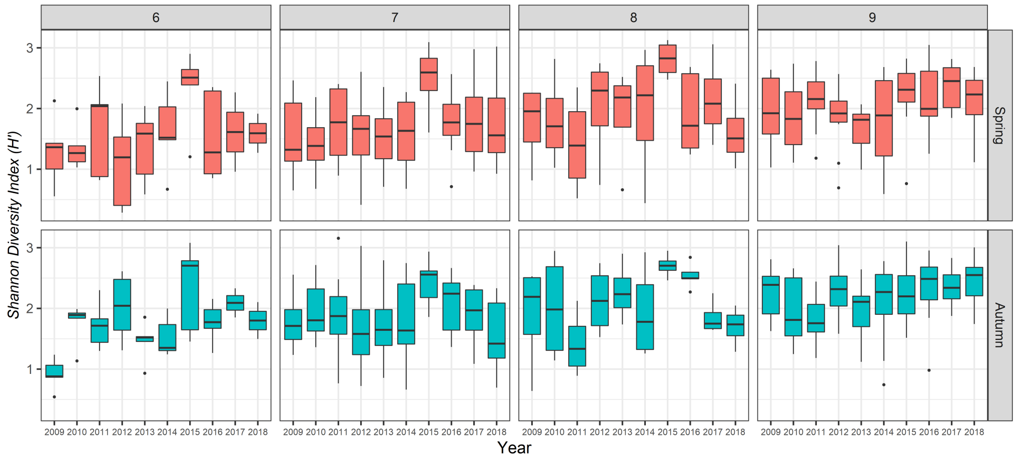


Figure 5.8: Distribution of Shannon Diversity Index (H') in the GOVD dataset from 2009 to 2018 by stream order and survey

is similar between the two surveys. Similar to the results of Species Richness and Species Evenness, it shows that sites with stream order 9 have a more stable and higher value in the index than in low stream ordered sites throughout the monitoring period.

In the temporal pattern of the index, sites with stream orders 8–9 have a clear downward movement in the trends at some point during 2010–2013 in both surveys and the extent of the change is more obvious in the spring survey than in the autumn survey. However, sites with stream orders 6–8 have no such distinct feature. These downward trends in sites with stream order 9 could be the effect of the construction or the completion of the barrages. In 2015, the peak in the index is identified in sites with stream order 6–8 in both surveys, but not in sites with stream order 9. These peaks are mainly found in sites located in the north region of the study area (e.g. M1–M4 and T2–T6b in Figure D.3) and appear to be related to the impact of the hydrological drought in 2015 which hit the north region hard. The response to the drought in Shannon Diversity Index is similar to those in the distribution of Species Richness and Evenness.

To summarise the findings, the application of the three diversity indices to the GOVD dataset displays the temporal change of diversities from 2009 to 2018 across the sites in the Nakdong river and its tributaries. The results show that the distribution of three diversities differs by stream order, mainly between sites with stream orders 6–8 and 9. During the whole period, sites with stream order 9 have a consistent increase in the number of species, while other sites fluctuate up and down, not having a monotonic direction in Species Richness. Species Evenness and Shannon Diversity Index in the catchment remain similar throughout the time period across all the sites with a few notable changes between 2010–2012 and in 2015.

The results of three diversity indices may be partially explained in connection with the construction of the barrages, and the hydrological drought in the catchment. Three diversity indices in all sites respond with the emergence of the peak in 2015 when the hydrological drought occurred in the catchment, but the response in sites with stream order 9 is relatively weak compared to other sites. In regards to the construction of the barrages, neither Species Richness nor Species Evenness in sites with stream order 9 shows any obvious change, whilst the results of Shannon Diversity Index display some degree of degradation in the index in the spring survey during 2011–2013 (the time of the construction and the completion of the barrages).

Although some degree of responses in the results of the three diversity indices appears to be related to the construction of the barrages and the hydrological drought, however, the results demonstrate that the application of the three diversity indices to the GOVD dataset does not greatly help to understand the changes in riverine environments appropriately in the Nakdong river and its tributaries. In particular, without the examination of the hydrological system in the Nakdong catchment, it would have been difficult to understand the meaning of the increase in 2015 in the results of diversity indices. Therefore, it is not beneficial to use diversity indices with diatoms alone in understanding changes in riverine environments.

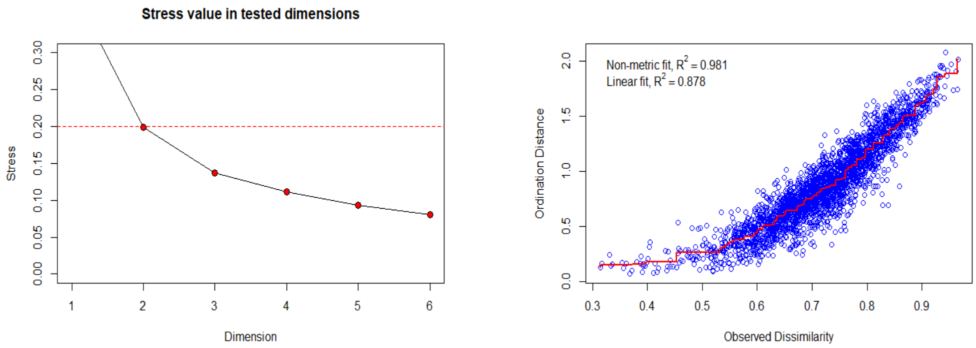
5.3.2 What variability found in multiple samples?

Neither the examination of the results of the mTDI nor the application of the diversity indices can provide a comprehensive picture of the responses of diatoms to the spatio-temporal change in water quality and environmental changes in the rivers such as the construction of the barrages and the hydrological droughts in the Nakdong catchment. This result suggests that there is a strong need to take a different approach in the application of diatoms to be able to assess river health as a holistic assessment.

Before moving on to taking a holistic approach with the GOVD dataset, variability in diatom assemblages from multiple samples within the government survey period was examined. Because the GOVD dataset consists of samples taken once per site during each period of two months in spring and autumn every year and variability within the survey periods is little known. Therefore, it is essential to examine how big variability in diatom assemblages is within a period and find out what environmental variable contributes to variability most. The examination of variability demonstrates what risks exist in the current sampling method and provides an insight into conceiving a better way to utilise diatoms in river health assessment in the future.

5.3.2.1 How big is variability in diatom assemblages?

In order to examine variability in diatom assemblages in multiple samples, the MYD dataset, consisting of two samples per site collected three weeks apart, was examined together with the ROD samples collected at the same time in the Nakdong river. Observations from the combined MYD and ROD datasets were projected in NMDS space after the three-dimensional space was determined through the dimensional check analysis with the Bray–Curtis dissimilarity method (Figure 5.9).



(a) Stress value at different dimensions (b) R^2 comparisons

Figure 5.9: Stress value and Stress plot to determine dimensions for the combined MYD and ROD datasets in NMDS analysis. Red dashed line in (a) indicates a threshold for stress value.

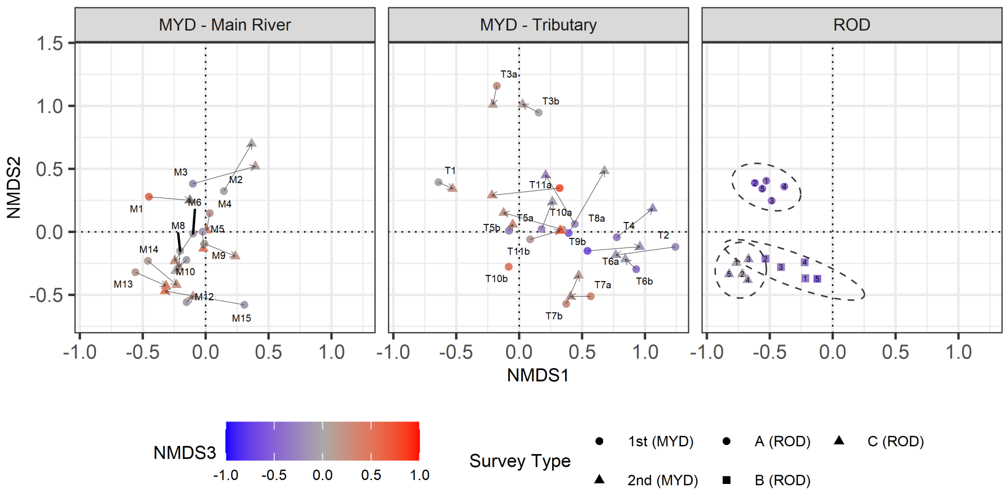


Figure 5.10: Observations of the MYD and ROD datasets in NMDS ordination plot. Dashed ellipses in ROD plot (right) indicate 95 % confidence level. All samples for the MYD dataset were sampled twice per site (1st and 2nd) except for T10b taken once. Dashed vertical and horizontal lines are arbitrarily drawn for ease of reading.

Figure 5.10 presents observations of two-time sampled diatom assemblages in the MYD dataset along with observations from the ROD dataset as a reference in the same NMDS ordination space. Despite the MYD samples taken twice three weeks apart in the rivers, observations in the main river and the tributaries are separated in NMDS space; samples from the main river are well-clustered with M1–M3 slightly distant from the rest, sitting on the left of the space, while the tributaries are located on the right and top sides of the space, broadly scattered by different rivers. For example, T1, T3a, and T3b are far away from the rest of the sites in the tributaries but located much closer to M1–M3 in the main river.

The visual inspection of Figure 5.10 shows that all sites in the MYD dataset have relocated themselves in NMDS space by moving different distances during the three weeks between 1st and 2nd samplings. Some sites such as T1, T3a, T3b, and T7a have two observations near with little movement, while the others like M1–M3 and M9b have significantly changed. Distance between the two samples per site is discussed in detail based on the segment length analysis in comparison with the observations in the ROD dataset later.

Figure 5.10 also exhibits that the location of M3, M10 and M14 in the MYD dataset is similar to the position of their counterparts Groups A–C in the ROD dataset in NMDS space despite the two-month gaps in sampling between the MYD and ROD datasets. Samples from both datasets are horizontally parallel on the NMDS2 axis, but different on the NMDS1 axis. For example, two M3 samples from the MYD dataset are positioned to the right side of Group A in the ROD dataset by 0.5 on the NMDS1 axis, while M10 and M14 are around 0.25–0.3 away from Group B and C on the NMDS1 axis. This similar structure between the two datasets suggests that the major relationship between the three sites is preserved from October to December despite the seasonal changes from autumn to winter.

The difference between the two samples in the MYD dataset and the difference between the observations in the ROD dataset are measured as segment length based on Euclidean distances in NMDS space. The results are presented by dataset and stream order (Figure 5.11).

Figure 5.11 (a) shows the mean distance between the observations by dataset. The mean distance between the two samples in the MYD dataset is 0.39 in the main river and 0.41 in the tributaries and the difference between the rivers is not significant. In contrast, the mean distance between the five samples within the group in the

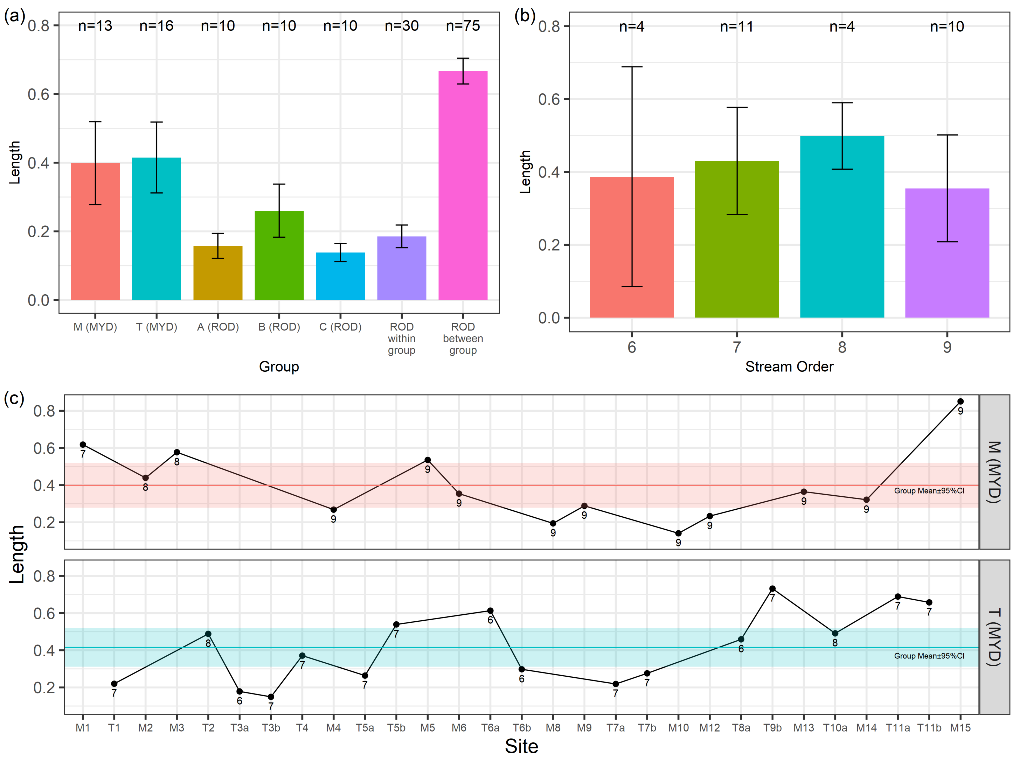


Figure 5.11: Comparisons of segment length in the MYD and ROD datasets. (a) Mean of segment length with 95 % confidence interval by group, (b) Mean of segment length for the MYD dataset by stream order, (c) Distribution of segment length between the two observations for the MYD dataset in the main river (M) and the tributaries (T). Numbers below points indicate stream order at site and horizontal solid line and coloured highlights represent group mean and 95 % confidence interval.

ROD dataset is significantly shorter at 0.2, while the average length between the group is over 0.6. These results mean that the difference in diatom assemblages between the two-time samples in the MYD dataset is bigger than the difference observed in the ROD samples taken at the same time within the group but smaller than the difference observed between the group in the ROD samples. In the MYD dataset, the segment length between the two observations varies by stream order (Figure 5.11 (b)). Mean distance in sites with stream order 9 is the shortest, while sites with stream order 8 are the longest. However, their difference cannot be considered statistically significant due to 95 % confidence levels overlapping.

The measurement of segment length between the observations in the MYD dataset is plotted in a longitudinal profile from upstream to downstream along with group means and a 95 % confidence interval to see their spatial patterns (Figure 5.11 (c)). It shows that three upstream sites M1–M3 and M15 in the main river move beyond the average during the three weeks, while sites in between them tend to change little. In the tributaries, the distribution of segment length is greatly different by sites rather than river. For example, T5b, T6a, T9b, and T11b have a longer segment length than the average, while T1, T3a, T3b, T6b, T7a and T7b are shorter in the length.

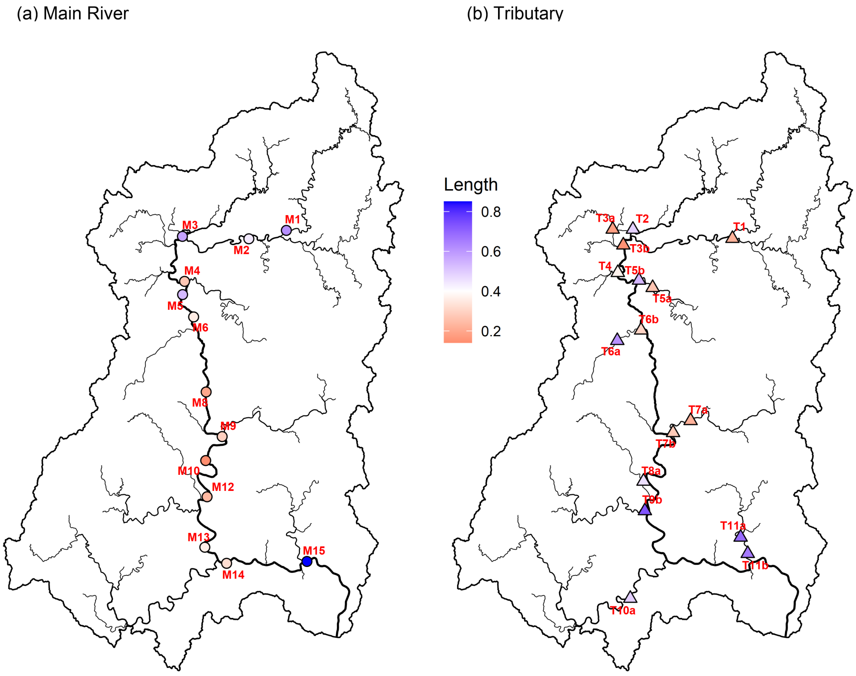


Figure 5.12: Geographical distribution of segment length between two observations per site in the MYD dataset

These results are geographically visualised on the map of the study area (Figure 5.12). The biggest changes are observed at upstream sites M1–M3 and downstream site M15 in the main river, whilst changes in between them are far below the average. Sites with little change are all located in the reaches regulated by the barrages in the main river, while sites with big changes are unregulated sections in the main river. In contrast, changes in the tributaries are geographically independent, although changes in T11a and T11b are almost big as in M15. For example, change at T1, geographically close to M1 is little compared to a big change at M1. In addition, T3a and T3b in tributary YG show relatively little change in the length, but adjacent T2 records above the mean in the length. The results also display that upstream and downstream sites within a river in the tributaries have different responses; in tributary WE, upstream site T5a changes little, while downstream T5b changes significantly, however, in tributary GM, upstream site T6a shows a big change compared to little change in downstream site T6b. This contrasting change within the same river in the tributaries means that changes in the tributaries are more likely to be caused by independent factors at the local scale, while changes in the main river appear to be related to the effect of the barrages.

To conclude this section, two observations per site sampled three weeks apart in the MYD dataset show that diatom assemblages have changed for that period. The difference in diatom assemblages between the two observations, expressed as the segment length, is not significantly different between the main river and the tributaries. However, it is bigger than the difference within the groups in the ROD samples taken at the same time but smaller than the difference between the groups in the ROD dataset. The distribution of segment length between the two observations in the MYD dataset is spatially different. Segment length in sites located in the regulated reaches by the barrages is relatively short, while other sites in unregulated reaches see a big change during the three weeks. In the tributaries, the distribution of segment length is geographically independent.

5.3.2.2 Comparisons of two longitudinal trajectories in the main river

Two observations per site sampled three weeks apart in the main river in the MYD dataset were subsetted and two longitudinal trajectories were made by connecting those observations by survey (1st and 2nd) in a longitudinal order from upstream (M1) to downstream (M15) in the main river. These trajectories then were compared in terms of trajectory shape and trajectory length as well as segment length between two consecutive observations in NMDS space.

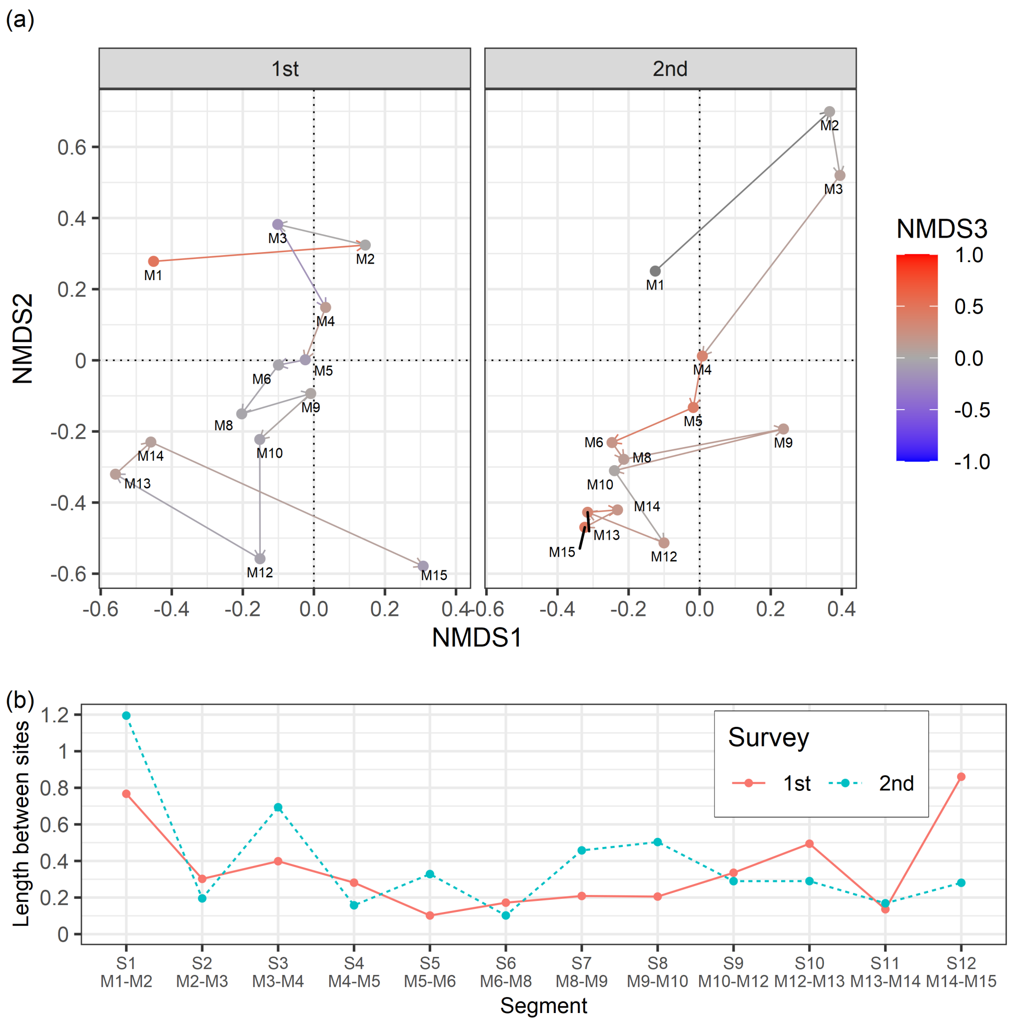


Figure 5.13: Comparisons of longitudinal trajectories in the main river by surveys for the MYD dataset. (a) Longitudinal trajectories of the main river by surveys in NMDS space. (b) Comparisons of segment length between consecutive sites in longitudinal trajectories. Dashed vertical and horizontal lines in (a) are arbitrarily drawn for ease of reading.

Figure 5.13 shows comparisons of two longitudinal trajectories by surveys in NMDS space and changes in segment length. Despite some sites having relocated themselves in the second sampling, the two longitudinal trajectories remain similar in shape (Figure 5.13 (a)). For instance, M1–M3, and M15 move significantly along the NMDS1 axis, nevertheless, the topology of segments such as M4–M6 and M8–M10 remains unchanged. This result is supported by a comparison of the two trajectories in trajectory length, which is defined as the sum of Euclidean distances between M1–M15 in NMDS space. Despite the significant changes in M1–M3, there is not much difference in length between the two. The length is 4.2 for the first trajectory, and 4.7 for the second trajectory. However, it is not possible to infer what this difference in length suggests because of the insufficient observations over time.

The two longitudinal trajectories were then broken down into segments which are defined as the Euclidean distance between two consecutive sites in the main river in ordination space. Those segments in two longitudinal trajectories were measured and compared in length to understand what changes in trajectory shape between the surveys mean.

Figure 5.13 (b) presents the distribution of segment length in the two longitudinal trajectories in the main river. As demonstrated above that the two longitudinal trajectories are similar in trajectory shape and trajectory length, they share similar features in the distribution of segment length. For instance, both surveys record a long segment in S1 (distance between M1 and M2), a short segment in S2 (M2–M3) and a rise again in S3 (M3–M4). Afterwards, they remain similar in segment length except for a long segment in S12 for the 1st trajectory. This similar pattern means that site M1 is different to sites M2–M3 in both surveys, while M2 and M3 are also different to the rest of the sites, moving together. Sites M4–M14 are different to those from the upstream of the main river by behaving similarly altogether.

The distributions of segment length in both surveys were tested to see if segments for 1st and 2nd surveys are from the same distribution using the Empirical Cumulative Distribution Function (ECDF) and the Kolmogorov—Smirnov (K–S) test. First, the visual inspection of the ECDF shows that they are likely from the same distribution (Figure 5.14). The result of the K–S test confirms that the distributions of segment length in both surveys are not from the normal distribution ($p < 0.0005$).

In summary for this section, despite the two observations per site in the main river scattered in NMDS space, the perspective of seeing them in a longitudinal

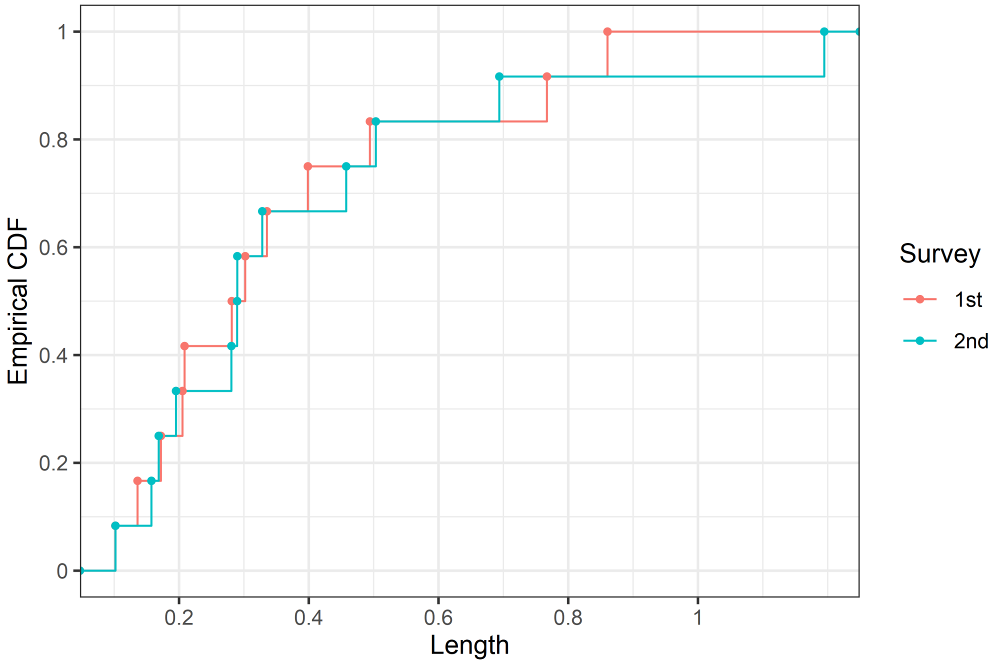


Figure 5.14: The Empirical Cumulative Distribution Function (ECDF) plot showing segment length in two longitudinal trajectories in the main river for the MYD dataset

order by surveys enables the formation of the two longitudinal trajectories from upstream to downstream and the comparisons of them in trajectory shape, length and segment length. The comparison of two longitudinal trajectories shows that despite the significant changes in some sites (M1–M3, and M15) between the two surveys, the two trajectories remain similar in shape and length. The breakdown of the longitudinal trajectories into segments shows that the distributions of segment lengths in both surveys are likely from the same distribution. These results mean that the overall diatom assemblages in the main river respond and behave similarly, maintaining the structure between the adjacent sites. These results are also in line with the result in the previous section that three groups in the ROD dataset sampled in the winter are parallel to their counterparts in the MYD dataset in the NMDS2 axis. In the meantime, the same analysis was not applied to samples from the tributaries due to the lack of observations within each river but the tributaries may present a similar result as in the main river.

5.3.2.3 What environmental variables contribute to variability?

The two observations per site sampled three weeks apart in the MYD dataset were paired with the measurements of seven environmental variables, obtained in situ on fieldwork. Those data then were used for Canonical Correspondence Analysis (CCA) to examine relationships between the environmental variables and sites and find the most influential variable in explaining variances between the two surveys. This analysis demonstrates which variable is the most responsible for the changes in the MYD dataset during the three weeks and provides an insight into how diatoms can be better used in assessing water quality moving forward.

The results of CCA show that the first axis explains 30.73 % and the second axis explains 20.59 % of variances in the MYD dataset (Figure 5.15 (a)). Regarding the environmental variables, the CCA1 axis is responsible for six out of seven variables, being positively correlated with DO, pH, ORP, TDS, and conductivity but negatively correlated with resistivity. Along the CCA2 axis, WT is the only variable which has a negative relationship.

Despite the CCA1 axis being the most important in explaining variances in the MYD dataset, during the two surveys, most sites are scattered along the CCA2 axis in CCA space with a clear shift of centroids to the top from the bottom (see the centroids in Figure 5.15 (a)) except for two samples from M1 that are significantly far away from the rest of the samples along the CCA1 axis (Figure 5.15 (b)). In the main river, most sites tend to remain at the centre of the space, clustered during the three week period, although M2 and M3 shifted to the top of the space in the second survey. However, this upward movement in the second survey is not the case for site M1. The second observation in M1 moved away even further to the left side from the rest of the samples along the CCA1 axis but hardly moved along CCA2 axis. In the tributaries, the extent of movement in ordination space between the surveys is greatly different by the site. For instance, T3a and T3b almost stayed in the same place and T6a and T6b moved down along the CCA2

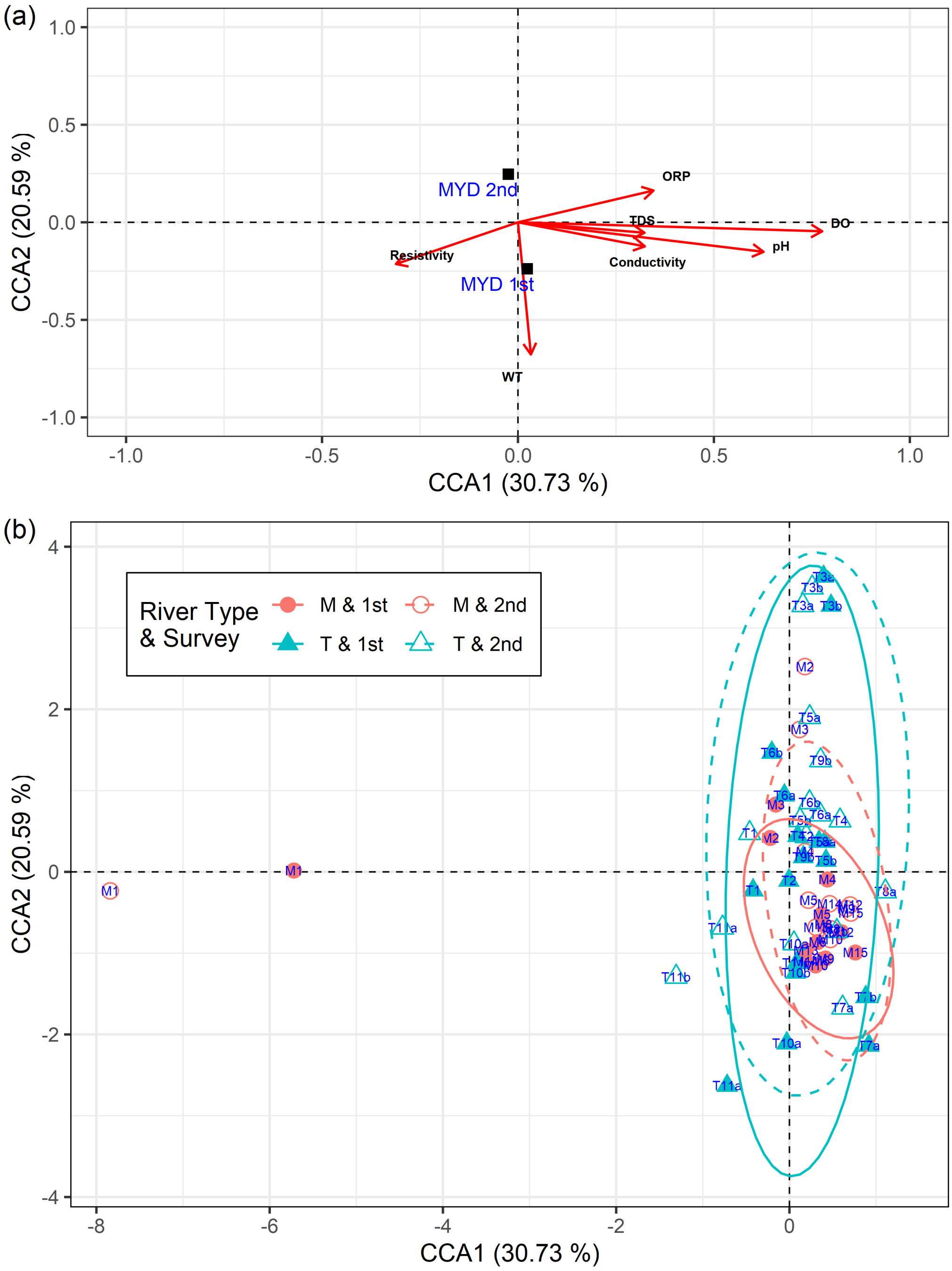


Figure 5.15: CCA biplots of the MYD dataset. (a) Ordination of environmental variables, (b) Observations (site scoring) by river type and survey. Square points in (a) indicates the centre points by surveys. Solid and dashed ellipses indicate 95 % confidence level.

axis, but their change was very little. However, some sites such as T1, T10a and T11a moved above their first-surveyed sample along the CCA2 axis in the space.

The results of CCA showing the observations during the two surveys are visualised to see their responses by stream order to the environmental variables (Figure 5.16). The classification of the results exhibits contrasting responses during the three week period across the sites by stream order. Sites with stream order 6–8 show relatively bigger changes along axis 2, while sites with stream order 9 barely change in any direction. For instance, sites with stream order 6 mainly moved along axis 2 with T8a moving along axis 1 and 2 at the same time. Most sites with stream order 7 tend to move along axis 2 in the space except for T4 and T11b which moved along axis 1 by a notable distance. Similarly, sites with stream order 8 also show a similar tendency with most sites moving along axis 2 during the same time period. In contrast, sites with stream order 9 remain similar in the space with very little movement.

This result means that the changes during the three weeks are mainly controlled by water temperature along axis 2 except for M1 site. It may also suggest that the response of diatom assemblages is different to water temperature by stream order in the rivers. However, with this gradient at stream orders 6–9, the difference by stream order is not likely significant. Instead, the difference may arise due to the effect of the barrages in the main river as all sites with stream order 9 are from the regulated reaches by the barrages in the main river. This means that those sites are more resistant to changes in water temperature due to the increased water level and water volume. This interpretation is in agreement with the least change in segment length in sites located in the reaches affected by the barrages (Figure 5.11).

The findings for this section are summarised as follows. During the three week period in autumn, the differences observed in two-time sampled diatoms are mainly driven by water temperature, not by pollution related-variables. The CCA results by stream order show that sites with stream order 9 have little change compared

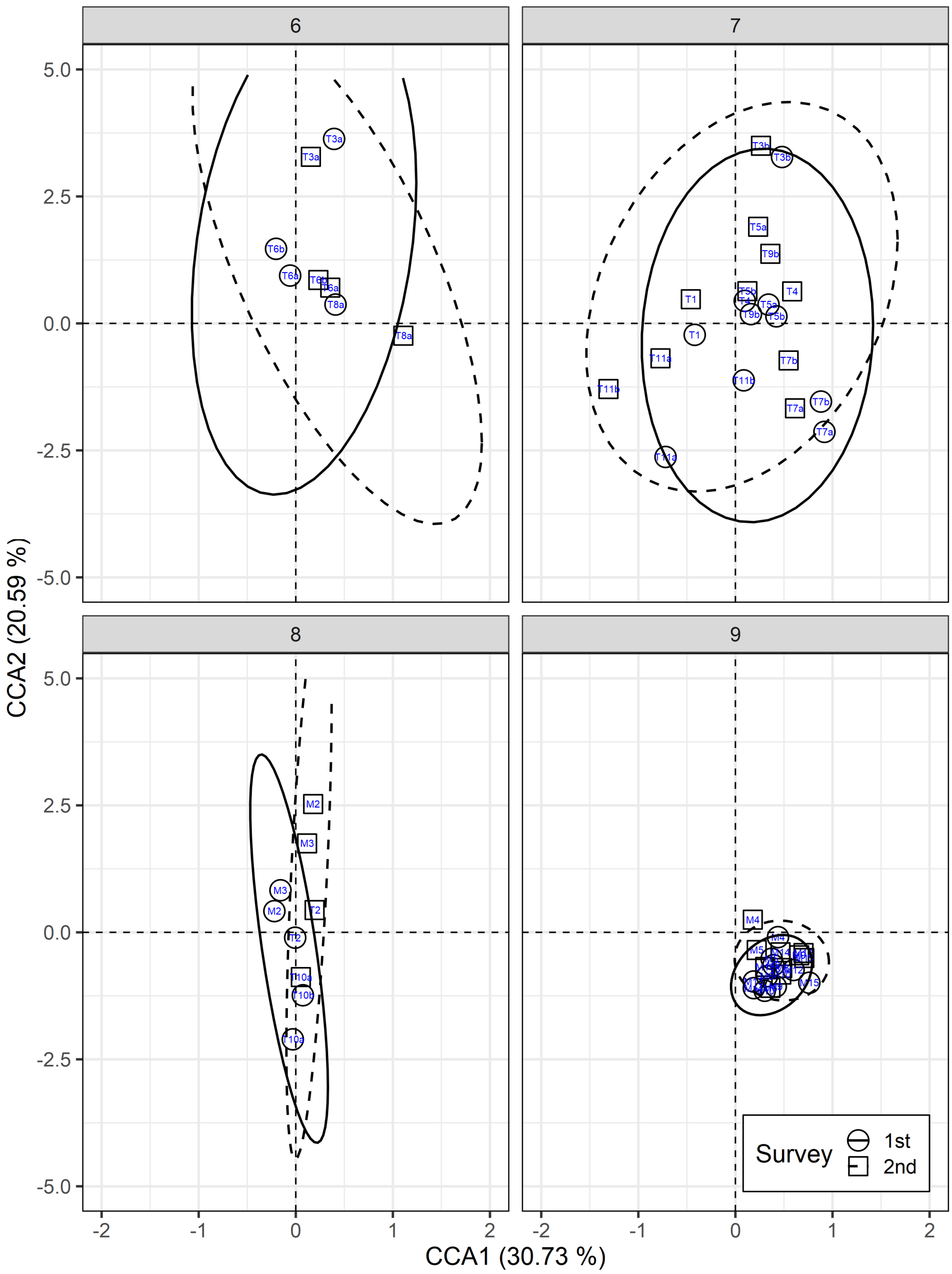


Figure 5.16: CCA plot of observations in the MYD dataset by stream order and survey. Two samples from M1 (Stream order 7) were excluded from display. Solid and dashed ellipses in (b) indicate 95 % confidence level.

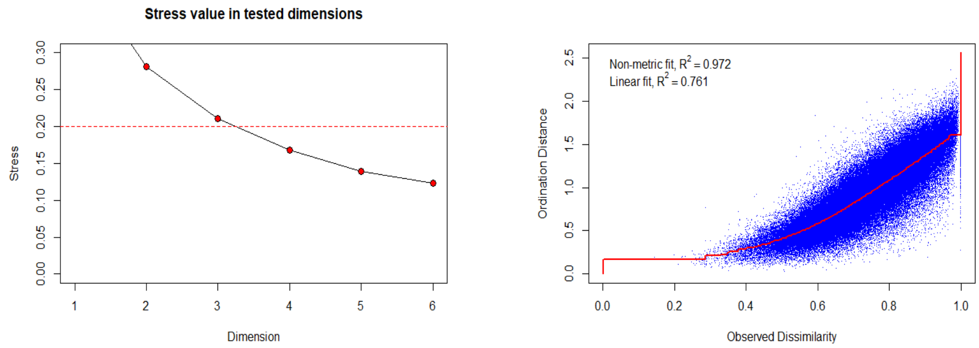
to sites with stream orders 6–8. This result does not mean that stream order is the factor controlling variance. It is more likely that because sites with stream order 9 are all from the reaches which are regulated by the barrages in the main river. This suggests that the barrages play a vital role in modifying the thermal system in the river through the increased water level and volume.

5.3.3 Distribution of the GOVD observations in NMDS space

As previously demonstrated in the results of the mTDI and diversity indices, there is a strong need to take a holistic approach in the application of diatoms to be able to understand the responses of diatoms for better river health assessment. This section examines changes in diatom assemblages in the GOVD dataset in the Nakdong catchment from 2009 to 2018 using NMDS analysis and provides the fundamental results for Community Trajectory Analysis (CTA).

To project the observations from the GOVD dataset in reduced dimensional space without misinterpretation of distances between them, dimensional space was determined through dimensional check analysis. The results confirm that the observations can be projected in four-dimensional space with a stress value of 0.167 (Figure 5.17 (a)), which is considered usable within a threshold (Clarke, 1993). In the selected space, 97.2 % of observations can be projected in non-metric fit (Figure 5.17 (b)) using NMDS analysis with Bray–Curtis dissimilarity method.

In examining the distribution of the GOVD dataset with various variables (year, site, survey and river type) in NMDS space, the first three-dimensional space was only selected to create NMDS ordination plots for the sake of convenience in visualisation. The other combinations of three-dimensional space (e.g. combinations of 1, 2, and 4 or 1, 3, and 4) were also reviewed before taking the decision, but they did not present any meaningful patterns (Figure D.4–D.5).



(a) Stress value at different dimensions (b) R^2 comparisons

Figure 5.17: Stress value and Stress plot to determine dimensions for the GOVD dataset in NMDS analysis. Red dashed line in (a) indicates a threshold for stress value.

5.3.3.1 Observations from the main river in NMDS space

Figure 5.18 presents longitudinal trajectories of diatom assemblages from M1 to M15 in the main river by the survey from 2009 to 2018. Despite the observations sampled each in spring and autumn, the trajectories in both surveys tend to move in the same direction in the space from the right to the left and from the middle to the bottom, and to the top throughout the time.

In detail, trajectories from the spring survey in 2009 and 2010 are mainly positioned on the first and fourth quadrants with a clear distinction between upstream and downstream sites in the NMDS3. In 2011, most sites slightly moved down along the NDMS2 axis with noticeable changes in the NMDS3 before they started to move to the top in 2012. During 2011–2012, changes in the NMDS3 are apparent in all sites with no clear division between upstream and downstream sites, but with a cluster of all sites nearby. The trajectory in 2013 moved to the left and centred around in the middle of the space. In 2014, with M1, M3, and M4 remaining in a similar position, the other sites moved to the top, resulting in stretching the trajectory. Afterwards, all sites tend to remain where they were with M1–M3 sites separated from the rest until 2017, although all observations in 2015 were well clustered. In 2017, the trajectory began to move downwards, maintaining its shape.

Trajectories in autumn have a similar tendency in movements over time. Until 2011, observations are located on the right side of the space with a clear separation between upstream and downstream sites in the NMDS3. In 2011, the trajectory moved downward to stretch further in length before they returned and centred around in the middle of the space with significant changes in the NMDS3 in 2012. With upstream sites M1–M4 separated from the rest, the overall trajectory moved upwards and to the left from 2013. Like the spring survey, in 2015, upstream sites M1, M3, and M4 moved up to form a cluster with the rest of the sites at a close distance. From 2017 onwards, the trajectory slowly moved downwards in space.

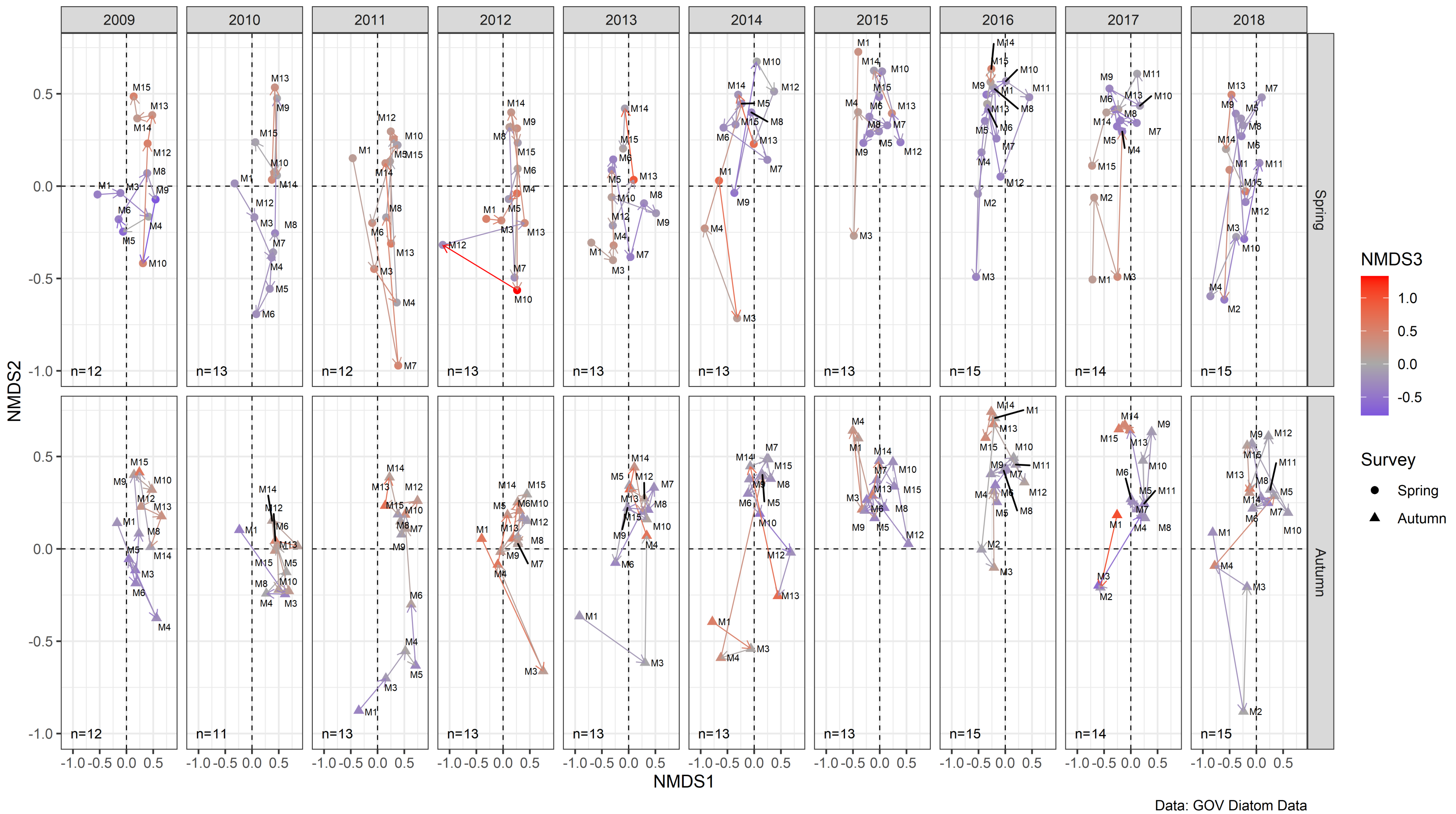


Figure 5.18: Longitudinal trajectories of diatom assemblages from M1 to M15 in the main river by survey; spring (upper) and autumn (lower). Dashed vertical and horizontal lines are arbitrarily drawn for ease of reading. The number of observations (n) varies by year.

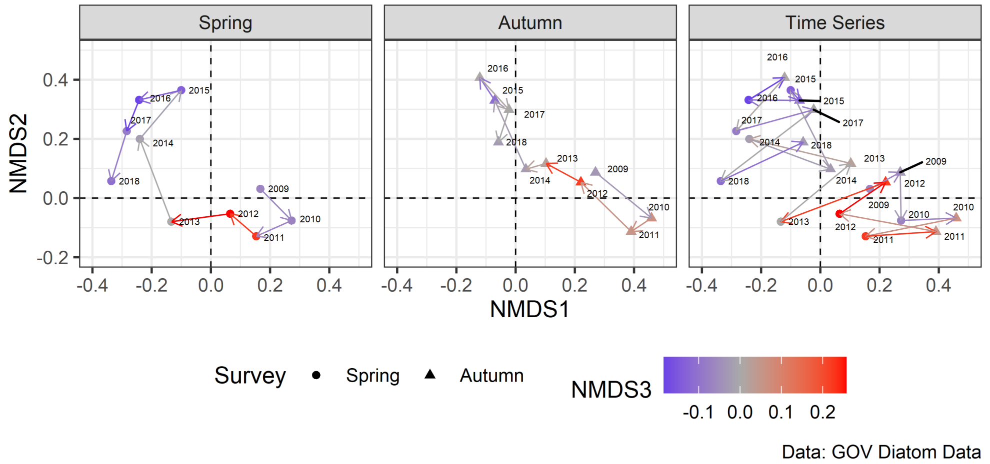


Figure 5.19: Movements of centroids for longitudinal trajectories in the main river over time. Survey basis plots (spring and autumn) show the movement of centroids by survey, while time-series plot (right) is a combined plot of them on time-series basis. Dashed vertical and horizontal lines are arbitrarily drawn for ease of reading.

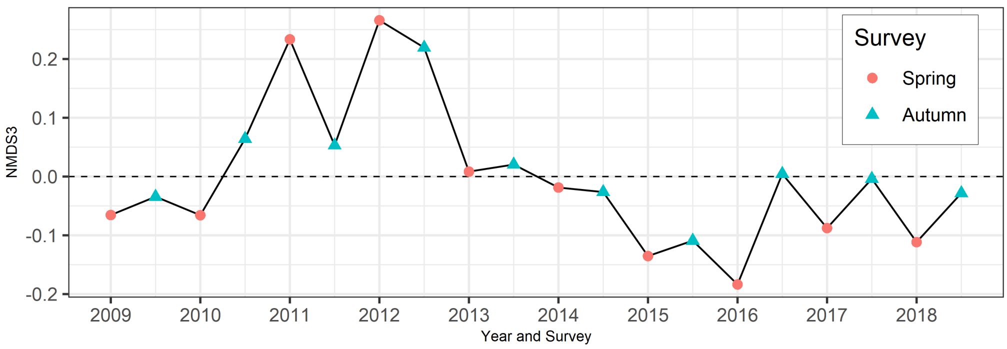


Figure 5.20: Movement of centroids for time-series trajectory in NMDS3. This graph is produced to help understand changes of centroids in NMDS3 for time-series trajectory in Figure 5.19.

The movement of trajectories in the main river over time is summarised in the plot which tracks down the movements of centroids for longitudinal trajectories on a survey basis as well as on a time-series basis (Figure 5.19). During a diatom monitoring period from 2009 to 2018, the centroid of each longitudinal trajectory does not remain in a similar position but constantly moves in space, accompanied by abrupt changes in the NMDS3 between 2011–2012.

In detail, the trajectories of centroids for spring and autumn are very similar in shape and direction (spring and autumn plots in Figure 5.19). Both have their first points for 2009 situated on the first quadrant of the space. Through 2011 and 2012, the centroids of the trajectories moved to the left and the top with remarkable changes in the NMDS3. The upward movement of centroids continued until 2015 for spring and 2016 for autumn. They then made a U-turn in the direction and slowly moved down in space.

In the meantime, the time-series trajectory of centroids (Time-series plot in Figure 5.19) that connects centroids for spring and autumn in chronological order provides a successive view on the movements of centroids in space. With the centroids for autumn always located to the right of spring points in space, the time-series trajectory creates a zig-zag pattern in the movements over time. In the overall movements of centroids, one notable feature is that centroids went through significant changes in the NMDS3 from 2011 to 2012 (Figure 5.20). From spring 2013, the movement in the NMDS3 considerably reduced, while the centroid abruptly started to shift to the top by a significant amount along the NMDS2 until 2015.

The distance between two centroids between spring and autumn in the same year considerably changes over time. Overall, centroids for spring and autumn within a year are always apart from each other in NMDS space, but the distance tended to increase from 2011 until 2015 when two centroids for spring and autumn were located at the closest distance in space. Afterwards, the distance started to increase again.

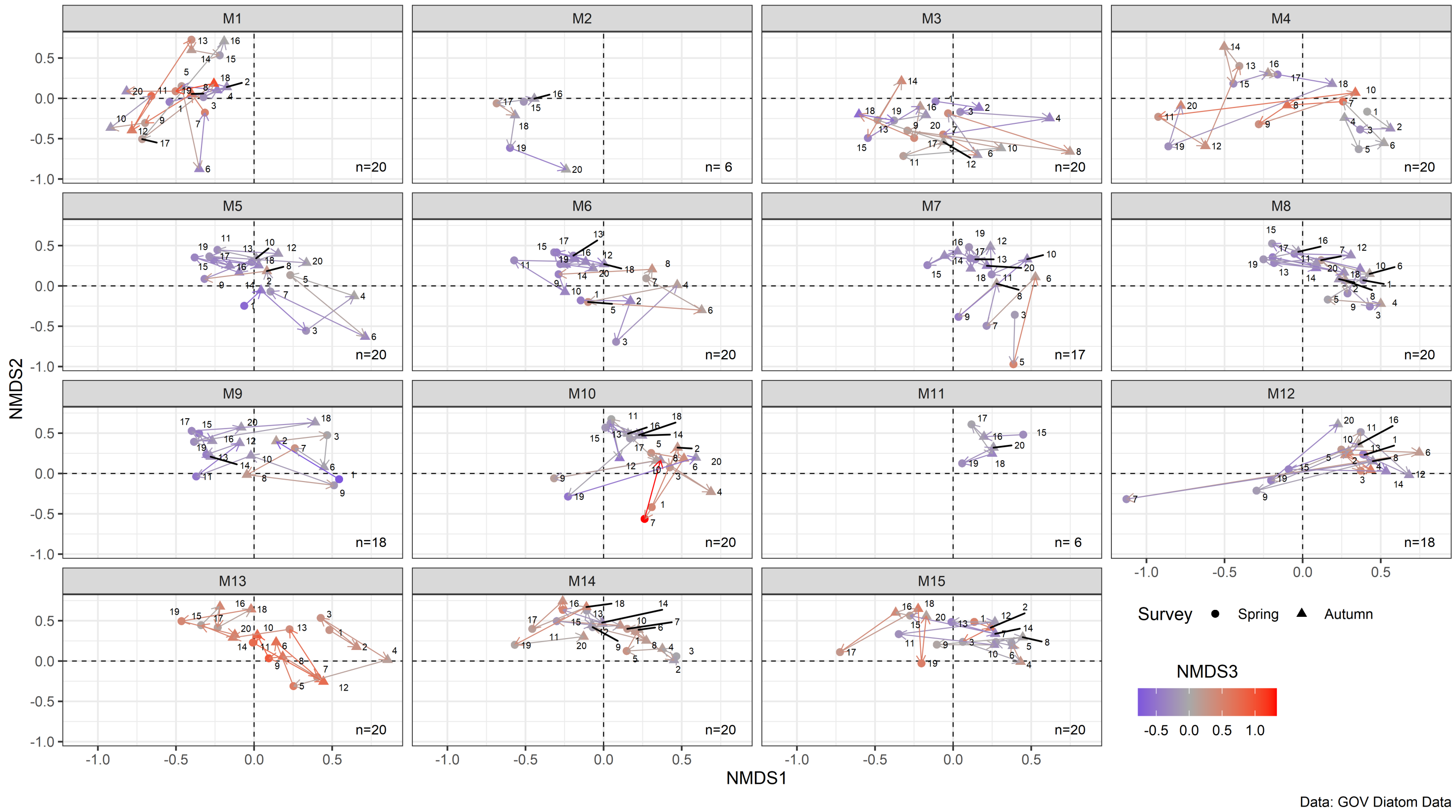


Figure 5.21: Time-series trajectories of diatom assemblages by the site in the main river. Stage is numbered in chronological order of survey. Odd number is sampled in spring, while even number is in autumn; 1-2009 Spring, 2-2009 Autumn, 3-2010 Spring, 4-2010 Autumn, 5-2011 Spring, 6-2011 Autumn, 7-2012 Spring, 8-2012 Autumn, 9-2013 Spring, 10-2013 Autumn, 11-2014 Spring, 12-2014 Autumn, 13-2015 Spring, 14-2015 Autumn, 15-2016 Spring, 16-2016 Autumn, 17-2017 Spring, 18-2017 Autumn, 19-2018 Spring, 20-2018 Autumn. Dashed vertical and horizontal lines are arbitrarily drawn for ease of reading.

Figure 5.21 shows the time-series trajectories of diatom assemblages by the site in the main river based on stage. The stage is numbered from 1 to 20 in chronological order of survey from spring in 2009 to autumn in 2018 (see stage number and corresponding year and survey in the caption for Figure 5.21). The trajectories in the main river can be roughly divided into four groups based on visual inspection of the direction and movement of the trajectory; three upstream sites (M1–M3), upper–midstream sites (M4–M9), midstream sites (M10–M12), and downstream sites (M13–M15).

The first group, comprising M1–M3, has no specific direction or movement in the time-series trajectory but keeps on moving randomly in space. Although M2 has a very limited number of observations, it is similar to the behaviour of M3 in the trajectory. In terms of changes in the NMDS3, M1 records intermittent changes at stages 7–14 and 18–19, while M3 goes through changes at stages 7–14. On the other hand, the second group from upper–midstream (M4–M9) has a trajectory moving from the fourth quadrant to the second quadrant over time. Observations at stages 1–6 tend to stay in the bottom right of the space, circling, but they tend to move to the top left of the space through stages 7–8, stabilising there. The third group (M10–M12) are similar to the second group in terms of position but they do not move to the left side of the space over time. Instead, M10 and M12 go through some changes in the NMDS3 at stages 1–7 and it is thought that M11 would have had similar changes at those stages. Lastly, the fourth group (M13–M15) has a trajectory characterised by a more horizontal shift in space by moving from the right to the left of the space.

The time-series trajectories of diatom assemblages in the main river (Figure 5.21) were further broken down to create the time-series trajectories by the survey and the site which exhibit how clustered observations are within each survey in NMDS space. The plots by the site for spring and autumn are presented in Appendix D (Figure D.6 and Figure D.7) and they clearly show different patterns of observations by the survey over time in the main river.

For example, in spring, sites at M3–M10 have two clusters of observations with the first one at stages 1–7 (9) and the second one at stages 9 (11)–19. These two clusters are separated in space with a fairly straight line. Sites M12–M14 have also a similar clustering pattern. However, site M1 at the most upstream does not. For the autumn survey, there is a weak or no clustering pattern observed in the time-series trajectories. Visually, sites M5 and M6 exhibit some extent of clustering with the first one made of stages 2–5 and the second one made of the rest of the observations, but overall the clustering pattern in autumn is not apparent as in spring. Also, the observations in autumn have moved in space, but the distance between them was relatively shorter than in spring.

5.3.3.2 Observations from the tributaries in NMDS space

Continuing to use the same results of NMDS analysis, the time-series trajectories of diatom assemblages in the tributaries are presented by rivers in Figure 5.22. Because of the limited number of observations considered in the tributaries, the longitudinal trajectories by rivers were not constructed. Instead, the time-series trajectories of diatom assemblages by the site were only examined.

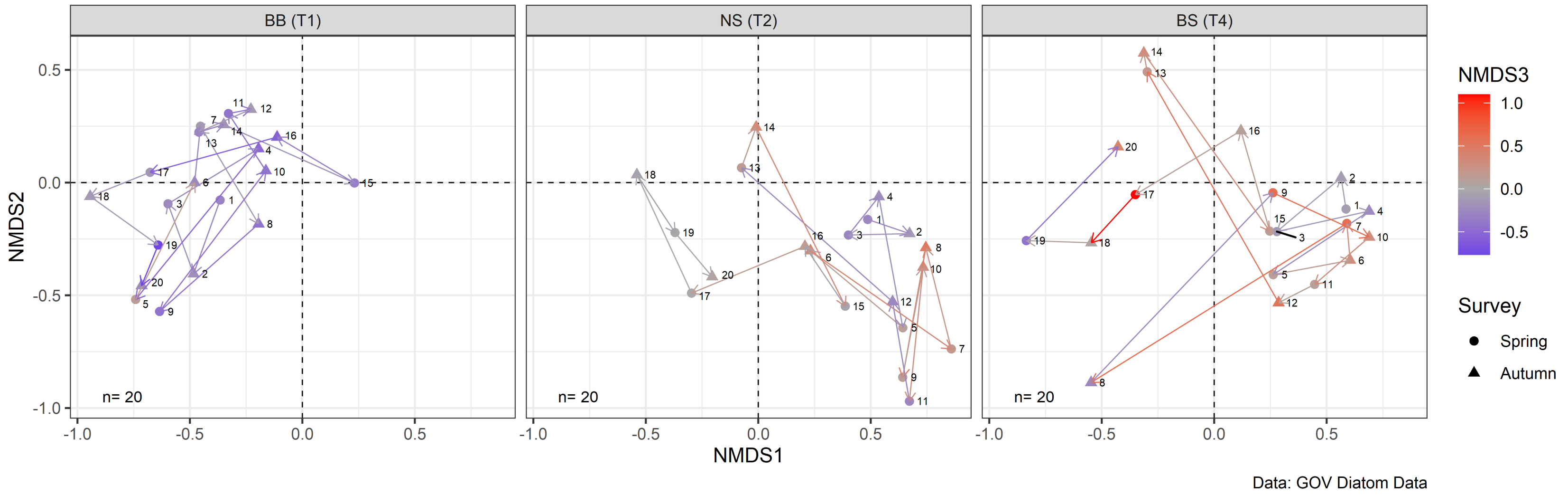


Figure 5.22: Time-series trajectories of diatom assemblages by the site in the tributaries (BB, NS, BS, YG, WE, GM, KH, HO, HG, NG and MR). When two sites were included in some tributaries, either a or b is added to site name. For example, tributary YG has two sites T3a located in upstream and T3b in downstream. However, the concept of upstream and downstream is relative and their distance to the main river is different. See their location in Figure 5.2. Stage is numbered in chronological order of survey. Odd number is sampled in spring, while even number is in autumn; 1-2009 Spring, 2-2009 Autumn, 3-2010 Spring, 4-2010 Autumn, 5-2011 Spring, 6-2011 Autumn, 7-2012 Spring, 8-2012 Autumn, 9-2013 Spring, 10-2013 Autumn, 11-2014 Spring, 12-2014 Autumn, 13-2015 Spring, 14-2015 Autumn, 15-2016 Spring, 16-2016 Autumn, 17-2017 Spring, 18-2017 Autumn, 19-2018 Spring, 20-2018 Autumn. The number of observations (n) varies by site and dashed vertical and horizontal lines are arbitrarily drawn for ease of reading.

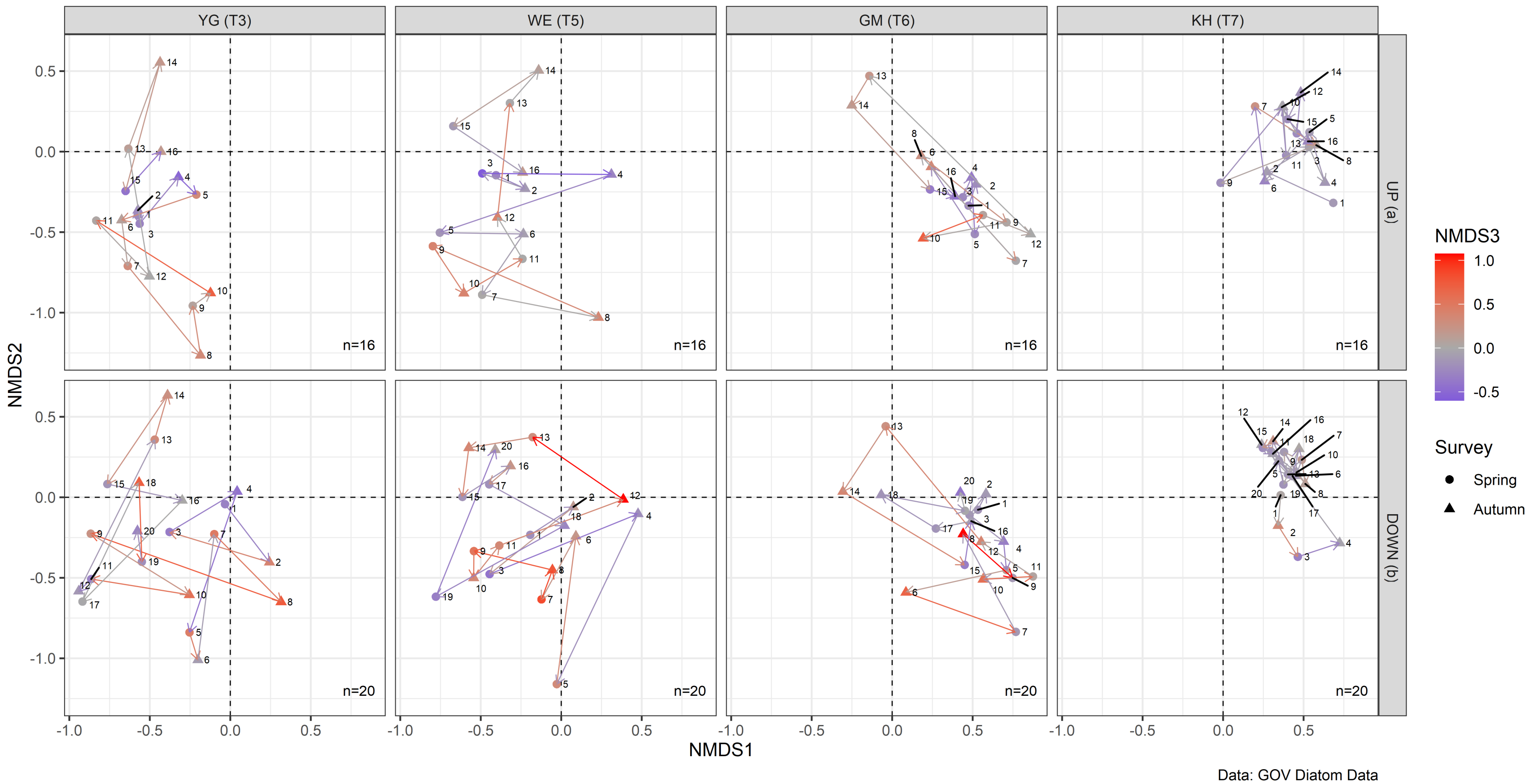


Figure 5.22: (continued)

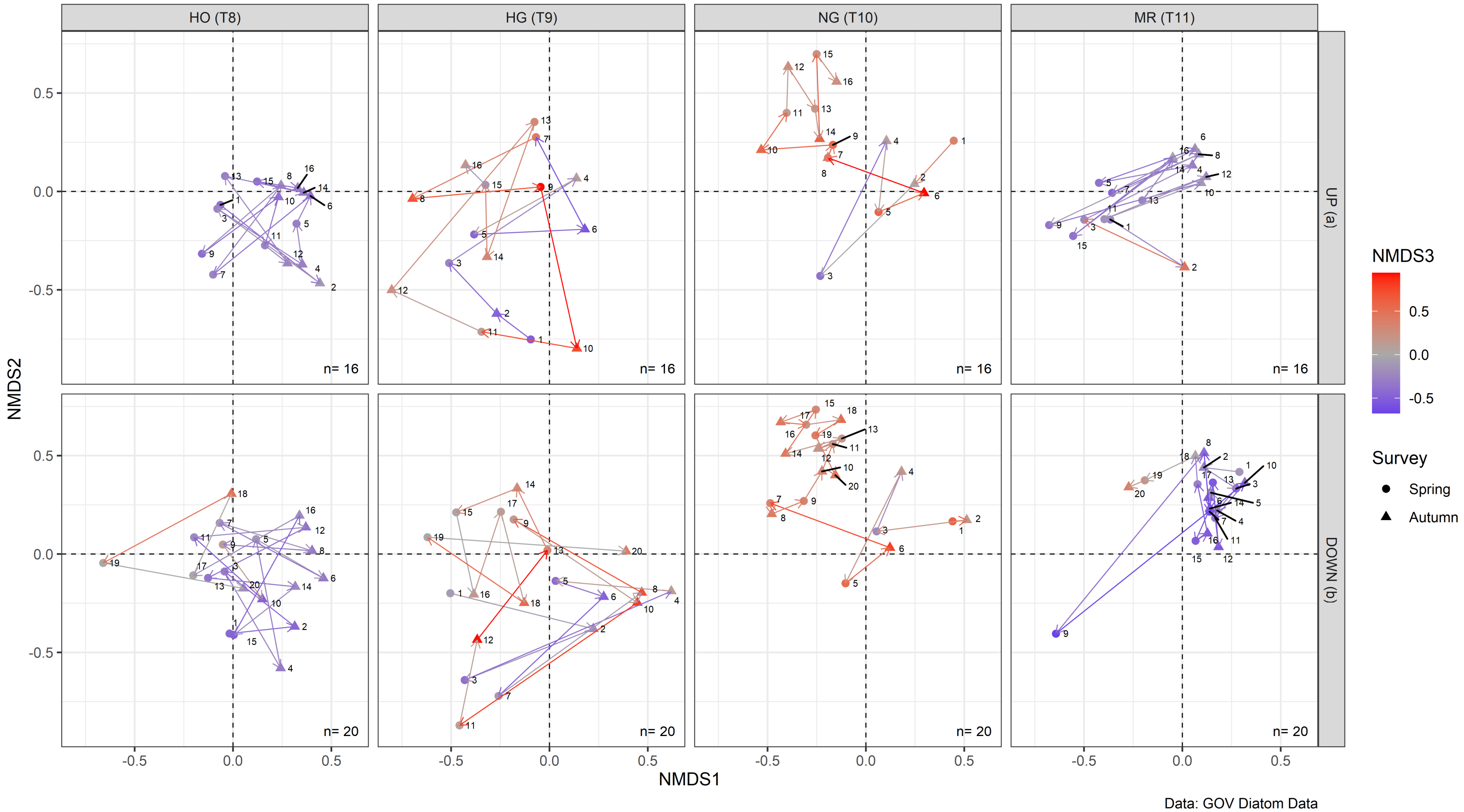


Figure 5.22: (continued)

Time-series trajectories in the tributaries differ greatly by the river and do not have the same consistent movement observed in the main river (Figure 5.22). Site T1 in tributary BB has a rotating trajectory on the left side of ordination space over time. There are occasions when the T1 trajectory makes fairly straight movement in line (e.g. during stages 15–18) but the overall tendency is that observations at site T1 have remained in a similar position. In contrast, sites T2 and T4 share interesting patterns in common in their trajectories. Both sites are similarly located on the right side of the space during stages 1–4, but after a period of disruptions at stages 5–12, the trajectories abruptly move upward at stages 13–14. Later, they also have divergent ends in the movement to the left side between stages 17–20.

Meanwhile, surprisingly, eight tributaries where one upstream site and one downstream site are included show a similar pattern of trajectories between upstream and downstream sites, unlike the expectation that diatom assemblages at downstream sites would experience more changes due to the effect of the barrages in the main river. Generally, the two sites at T3 and T5 (Tributaries YG and WE) have a similar trajectory in position and pattern, mainly sitting on the left side of the space. Both rivers also go through significant changes in the NMDS3 through stages 5–12 with a considerable movement to the top at stages 13–14. Next, T6a and T6b in tributary GM have trajectories on the right side of the space, behaving similarly to each other over time. They also have observations at stages 13–14 that are distinctive in the trajectories.

Tributary KH has very densely clustered observations in the trajectories at both upstream and downstream sites, while tributary HO has observations randomly moving around at the centre of the space. Similarly, two trajectories in tributary HG have centred around in a similar position with some changes in the NMDS3 at stages 7–14. Tributary NG has two trajectories sitting at the top of the space in both sites. In contrast, tributary MR has different trajectories in upstream and downstream sites with some degree of consistent movement. For instance, upstream site T11a shows a consistent switch of observations between spring and autumn in the trajectory, while downstream site T11b shows a weak switching pattern with little difference between the two surveys.

To summarise, the time-series trajectories in the tributaries show that observations in the tributaries have consistently moved in the space during 2009–2018, but the trajectories do not share the same pattern of movement observed in the main river. Instead, they tend to hover around at the centre of their usual positions in ordination space. Furthermore, one notable feature observed in the trajectories is that rivers in the north of the study area (Tributaries NS, BS, YG, WE and GM) have distinctive upward movements to the top of the space coincidentally found at stages 13–14. However, these patterns are not observed in other tributaries in the south or are very weakly visible in tributary HG.

5.3.4 Trajectory analysis

In the previous analysis, the observations of the GOVD dataset in NMDS space that form trajectories on a time-series basis (chronological order using stages 1–20) or a survey basis (spring and autumn) were visually examined by the site in the main river as well as in the tributaries.

In this section, those observations and trajectories in NMDS space continue to be used for Community Trajectory Analysis (CTA). The CTA is a framework, presented by De Cáceres et al. (2019) to compare community trajectories based on geometric analysis such as geometric properties of a community trajectory, projections of a community state onto a community trajectory, convergence/divergence between a pair of community trajectories, geometric resemblance between a pair of community trajectories, and spatial variation in community dynamics. The application of this framework to the results of NMDS analysis facilitates comparing the changes of diatom assemblages over time in the form of trajectory length and angle. Furthermore, the results of this analysis help understand the responses of diatom communities to seasonal changes, other environmental factors such as hydrological droughts, and other disturbances such as the construction of the barrages.

Thus, trajectory length and trajectory angle for the time-series trajectory and the survey-based trajectory were analysed by the site in the main river and the tributaries using two categorical variables (stage and survey). In this analysis, instead of using the nominal division (the main river and the tributaries), the Strahler stream order was adopted as a variable because stream order indicates a level of branching in the river system. It is related to physical characteristics in rivers as conceptualised in the River Continuum Concept (Vannote et al., 1980). It is also worth noting that sites at the reaches of the main river between the barrage 1 and the barrage 8 (see the locations in Figure 5.2) have all stream order 9. Stream order for all sites is presented in Table 3.1.

In this section, the analyses of trajectory length and trajectory angle were carried out on a basis of two orders; a time-series order and a survey-based order. First, based on a time-series order, Time-series Segment Length (TSL) and Time-series Segment Angle (TSA) were measured between observations in the time-series trajectories in the main river and the tributaries (Figure 5.21 and 5.22). Based on a survey-based order, Survey-based Segment Length (SSL) and Survey-based Segment Angle (SSA) were obtained using the survey-based trajectories (Figure D.6 for the Spring and Figure D.7 for the Autumn survey). The analysed geometric features for segment length and angle are defined in Table 5.1.

5.3.4.1 Distribution of Trajectory Length

Trajectory Length (TL_{site}) is defined as the total distance that each site has travelled in NMDS ordination space during ten years from stage 1 to 20 in the time-series trajectories of diatom assemblages in the main river (Figure 5.21) and the tributaries (Figure 5.22). This is calculated as the sum of the Time-series Segment Length (TSL) between two points in the time-series trajectories in the space using the equation 5.3.

Table 5.1: Definitions of types of segment length and segment angle

Year	Survey	Stage	Time-series Segment Length (TSL)	Time-series Segment Angle (TSA)	Survey-based Segment Length (SSL)	Survey-based Segment Angle (SSA)
2009	Spring	1	-	-	-	-
2009	Autumn	2	TSL2	-	-	-
2010	Spring	3	TSL3	TSA3	SSL2010 Spr	-
2010	Autumn	4	TSL4	TSA4	SSL2010 Aut	-
2011	Spring	5	TSL5	TSA5	SSL2011 Spr	SSA2011 Spr
2011	Autumn	6	TSL6	TSA6	SSL2011 Aut	SSA2011 Aut
2012	Spring	7	TSL7	TSA7	SSL2012 Spr	SSA2012 Spr
2012	Autumn	8	TSL8	TSA8	SSL2012 Aut	SSA2012 Aut
2013	Spring	9	TSL9	TSA9	SSL2013 Spr	SSA2013 Spr
2013	Autumn	10	TSL10	TSA10	SSL2013 Aut	SSA2013 Aut
2014	Spring	11	TSL11	TSA11	SSL2014 Spr	SSA2014 Spr
2014	Autumn	12	TSL12	TSA12	SSL2014 Aut	SSA2014 Aut
2015	Spring	13	TSL13	TSA13	SSL2015 Spr	SSA2015 Spr
2015	Autumn	14	TSL14	TSA14	SSL2015 Aut	SSA2015 Aut
2016	Spring	15	TSL15	TSA15	SSL2016 Spr	SSA2016 Spr
2016	Autumn	16	TSL16	TSA16	SSL2016 Aut	SSA2016 Aut
2017	Spring	17	TSL17	TSA17	SSL2017 Spr	SSA2017 Spr
2017	Autumn	18	TSL18	TSA18	SSL2017 Aut	SSA2017 Aut
2018	Spring	19	TSL19	TSA19	SSL2018 Spr	SSA2018 Spr
2018	Autumn	20	TSL20	TSA20	SSL2018 Aut	SSA2018 Aut

* TSL is numbered by stage of the endpoint. For example, TSL2 is the segment length between stages 1 and 2.
* TSA is numbered by stage of the endpoint. For example, TSA3 is the segment angle between TSL2 and TSL3.
* SSL is numbered by year and survey of the endpoint. For example, SSL2010 Spr is the segment length between stages 1 and 3.
* SSA is numbered by year and survey of the endpoint. For example, SSA2011 Spr is the angle between SSL2010 Spr and SSL2011 Spr.

$$TL_{\mathrm{site}} = \sum_{i=2}^{20} TSL_{\mathrm{n}} \tag{5.3}$$

where TSL_{n} (Time-series Segment Length), defined as the distance between two points in time-series order in NMDS space, is obtained by the equation 5.4 below.

$$TSL_{\mathrm{n}} = \sqrt{(x_n - x_{n-1})^2 + (y_n - y_{n-1})^2 + (z_n - z_{n-1})^2} \tag{5.4}$$

where n is stage and $2 \leq n \leq 20$.

Figure 5.23 exhibits TL_{site} across the site for a period from spring 2009 (stage 1) to

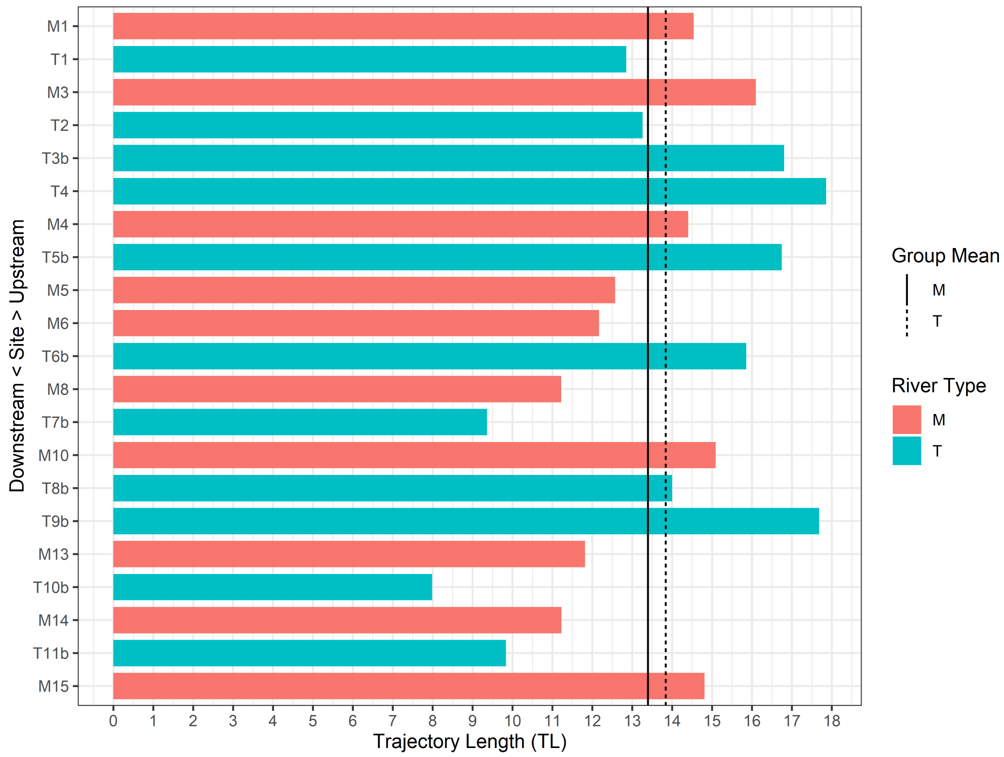


Figure 5.23: Distribution of Trajectory Length (TL_{site}) by the site between stages 1–20. Sites with the missing surveys were removed for the visualisation and group mean calculations (M2, M7, M9, M11, and M12 in the main river; T3a, T5a, T6a, T7a, T8a, T9a, T10a, and T11a in the tributaries).

autumn 2018 (stage 20) in the study area. Sites, where the number of the survey is below 20, were excluded for the visualisation and group mean calculations (see caption in Figure 5.23). During the time period, the mean of TL is 13.39 in the main river, and 13.83 in the tributaries, and the difference is not significant.

The distributions of TL in the main river and the tributaries are different. In the main river, TL in the upstream sites tends to be longer than in the mid- and downstream. For example, TL_{M1}, TL_{M2}, and TL_{M4} are above the average, while TL_{M5}–TL_{M14} are below the average. In the tributaries, there is a great variation of TL with TL_{T3b}, TL_{T4}, TL_{T5b}, TL_{T6b}, TL_{T9b} some distance above the average and TL_{T7b}, TL_{T10b}, TL_{T11b} some distance below the average.

Overall, the examination of the distribution of TL shows that trajectory length at a site differs greatly with a variation. However, it is not enough to draw any meaningful interpretation in terms of the responses of diatom communities between the main river and the tributaries to seasonal changes, other environmental factors and other disturbances in the catchment. Also, due to the incomplete number of the survey, the exclusion of 13 sites out of 34 makes it difficult to compare their responses with others.

Accordingly, *TL* is further broken down to *TSL*. TSL_{n} is the segment length in NMDS space between two observations at stages n and n-1 in *TL*, and their consecutive changes between stages throughout the time show the responses of diatom communities through seasonal transitions. The hypothesis is that change of TSL_{n} between observations would be similar through constant seasonal transitions unless there are any substantial changes in the environment. The distribution of TSL_{n} and its mean over time by stream order are visualised in Figure 5.24 and Figure 5.25.

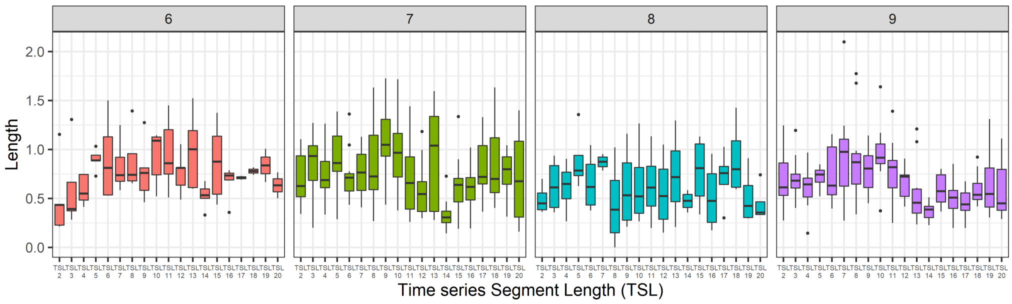


Figure 5.24: Distribution of Time-series Segment Length (TSL_{n}) during stages 1–20 by stream order

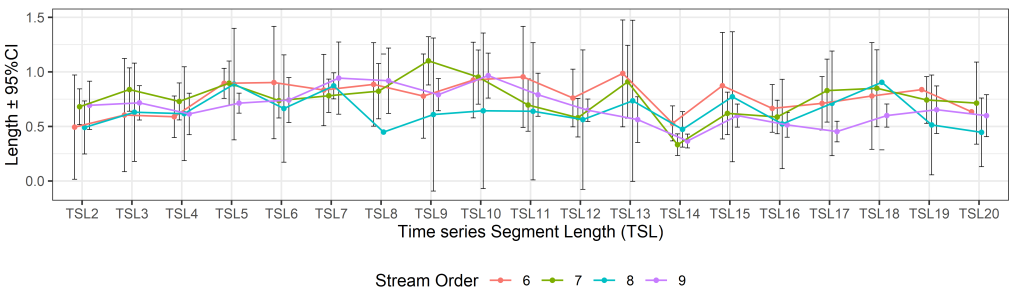


Figure 5.25: Temporal changes in the mean of Time-series Segment Length (TSL_{n}) during stages 1–20 by stream order. Error bars represent 95 % confidence interval.

Figure 5.24 shows that changes of TSL_n greatly differ throughout the time by stream order. This means that unlike the hypothesis, the responses of diatom communities have not been steady across the site but substantially different. For example, sites with stream order 6 have a fairly stable distribution of TSL over time within a range of 0.5–1.0 in length. Similarly, sites with stream orders 7 and 8 have a relatively steady pattern with occasional ups and drops including the significant drop at TSL_{14}. In contrast, sites with stream order 9 have a clear up and down pattern in the distribution of TSL with some extreme values concentrated during a high period in TSL_7–TSL_{11}. Particularly, they have a significant increase in the length during TSL7–TSL11 which is a period from autumn 2011 to spring 2014 (see Table 5.1 for periods) before the length gradually comes down.

One common feature in the distribution of TSL is a sudden drop in the length at TSL_{14} across all sites in all stream orders. TSL_{14} is the measure of the changes between spring and autumn in 2015, and the shortest length in NMDS space means that the diatom assemblages at most sites had changed little from spring to autumn in 2015. This simultaneous fall at TSL_{14} across all sites is more clearly observed in the mean changes of TSL_n (Figure 5.25). Figure 5.25 shows that throughout the time period, there has been little change in the mean of the length in sites with stream orders 6–7 except for the drop at TSL_{14}, while sites with stream order 8 have a fall at TSL_8 after a smooth rise. Sites with stream order 9 have gradually decreased in the length after a high period during TSL_7–TSL_{10} that are the changes between autumn 2011 and autumn 2013 (see Table 5.1 for periods).

Similarly, Survey-based Segment Length (SSL) between two observations in two consecutive years in the same survey was calculated based on the survey-based trajectories in the main river and the tributaries. The temporal change of SSL shows the timing of changes in diatom assemblages. The rationale for this analysis is that samples taken at the same survey period would be positioned nearby in NMDS space, maintaining a similar composition of diatom assemblages. There are two types of SSL depending on survey; SSL_{year}Spr and SSL_{year}Aut. They were obtained by the equation 5.5 below.

$$SSL_{\mathrm{y}}Spr/Aut = \sqrt{(x_n - x_{n-2})^2 + (y_n - y_{n-2})^2 + (z_n - z_{n-2})^2} \qquad (5.5)$$

where y is year of survey when n is stage number and $3 \leq n \leq 20$.

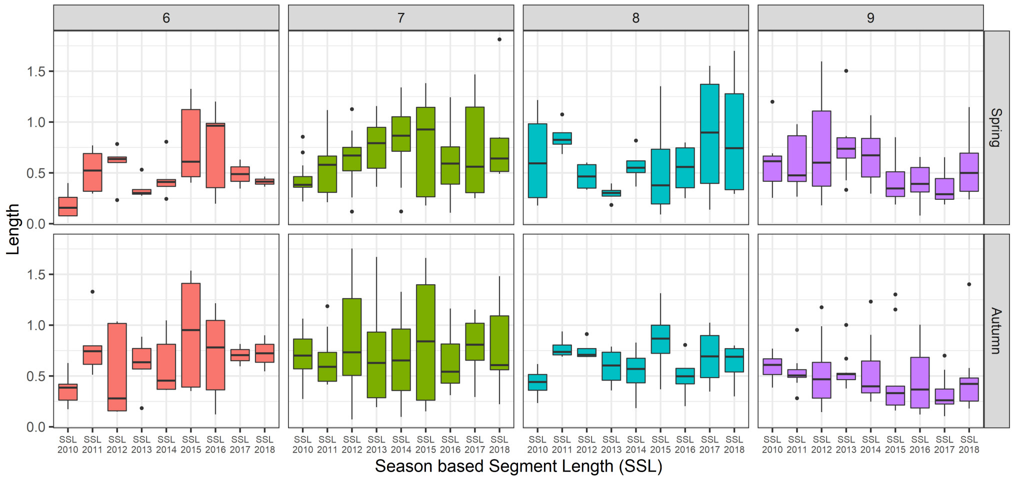


Figure 5.26: Distribution of Survey-based Segment Length (SSL_{y}) during stages 1–20 for spring (upper) and autumn (lower) by stream order

The distributions of SSL_{y}Spr and Aut by stream order are presented in Figure 5.26. The length of SSL is the measure of changes between two consecutive observations of diatom assemblages in the same survey, and the shorter it is, the more similar they are. Overall, in comparison within the survey, SSL in both surveys considerably fluctuates over time with some variations by stream order and year. For example, in sites with stream orders 6–8, SSL_{2015}Spr and SSL_{2015}Aut are commonly high. This means that compared to the observations in 2014, diatom assemblages sampled in 2015 for both surveys have substantially changed, being far away from the usual position in NMDS space. However, this signal is not observed in sites with stream order 9. Instead, sites with stream order 9 have an increase in SSL in both surveys between 2012–2013. This means that diatom assemblages sampled in both surveys in 2012 and 2013 have moved away from the previous location in NMDS space by a significant distance. This rise is more distinctive in spring than in autumn.

Another notable feature in SSL is that sites with stream order 9 clearly diminish in SSL and record a length of below 0.5 with little variation after SSL_{2014}. These changes in the length are small compared to any other length in all sites. This indicates that observations in sites with stream order 9 become similar to each other in diatom assemblages. However, these changes are not identified in other sites with stream orders 6–8.

5.3.4.2 Distribution of Trajectory Angle

Trajectory Angle is defined as an angle between two consecutive segments in NMDS space. The angle is measured from the direction of the first segment to the direction of the second segment and can mathematically range from 0° to 180° when two segments are straight or reversed. Like the distribution of trajectory length, two types of trajectory angles, Time-series Segment Angle (TSA_{n}) and Survey-based Segment Angle (SSA_{y}) were analysed using the time-series trajectories and the survey-based trajectories.

First, the Time-series Segment Angle (TSA_{n}) in the time-series trajectories was measured. TSA is the angle formed between two time-series segments (TS) made of three consecutive observations in NMDS space and was obtained by the equation 5.6 below.

$$TSA_{\mathrm{n}} = TS_{\mathrm{n\text{-}1}} \angle TS_{\mathrm{n}} \tag{5.6}$$

where n is stage and $3 \leq n \leq 20$.

Figure 5.27 is a histogram showing the frequency of Time-series Segment Angle (TSA_{n}) in the GOVD observations by stream order. Theoretically, the angle of 180° in the results indicates a return to exactly where it was a year ago, but there is no such angle identified at all. Instead, the results show that the most frequent angles found in TSA_{n} are obtuse at 120–160° across all sites, while acute angles are very rare. Obtuse angles in the time-series trajectories mean that the

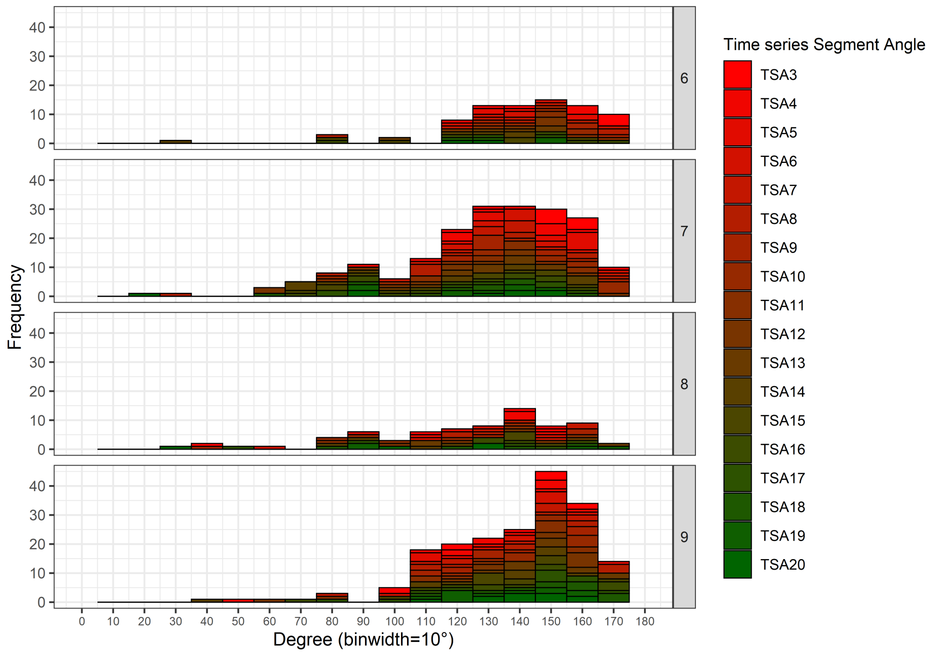


Figure 5.27: Histogram showing the frequency of Time-series Segment Angle (TSA_{n}) by stream order (Binwidth=10°)

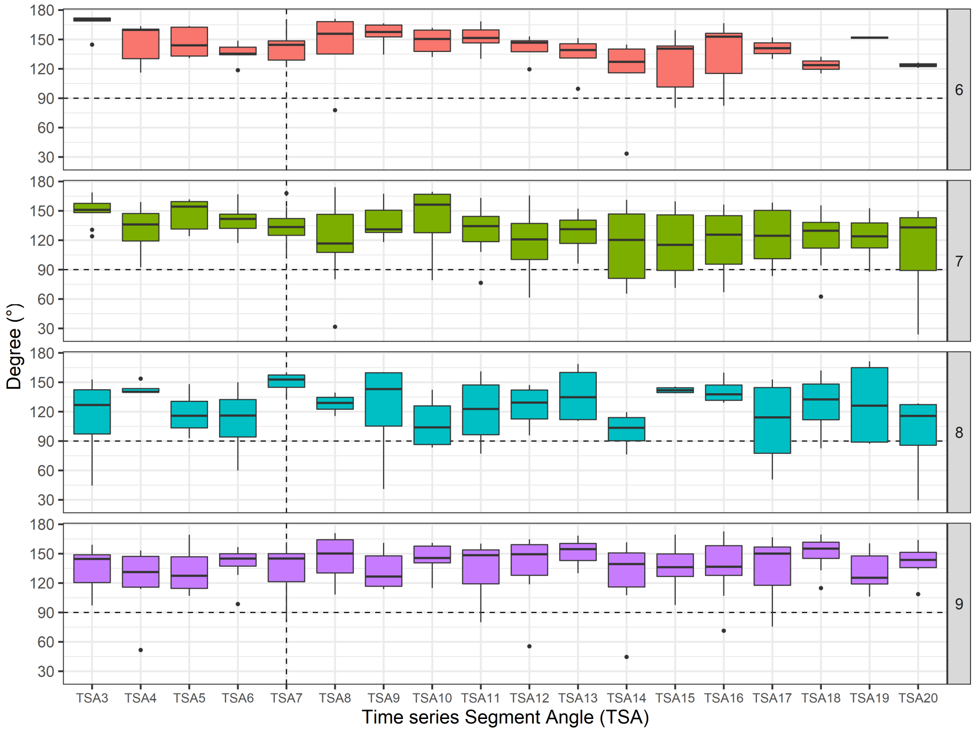


Figure 5.28: Distribution of Time-series Segment Angle (TSA_{n}) by stream order. Horizontal dashed lines indicate right angle and vertical dashed line indicates TSA_7, which is the angle from autumn 2011 to Spring 2012 when the construction of the barrages in the main river was completed in April 2012.

third observation takes a hard turn from the direction of the segment between the first and the second observations in NMDS space and tends to come back close to where the first observation was. In the time-series trajectories, the first and the third observations are sampled in the spring survey, while the second and the fourth observations are sampled in the autumn period. In the case of right or acute angles, the third observation continues to go straight or diagonally move far away from the first observation in NMDS space.

With these in mind, Figure 5.28 shows the dominance of obtuse angles in the distribution of TSA_{n} during the whole monitoring period. Particularly, sites with stream order 9 show consistency in the angle over time in comparison with other sites with stream orders 6–8. Provided that two notable changes in TSL_7 and TSL_{14} are identified in the results of Time-series Segment Length (Figure 5.24), it is worthwhile to have a look at the angles corresponding to those in the length. The results of TSA_{n} (Figure 5.28) show that TSA_7 do not have any different angle in particular. TSA_7 is made up of three observations from spring 2011 to spring 2012 via autumn 2011 and indicates the direction of change in diatom assemblages when the barrages were completed in April 2012. However, the dominance of obtuse angles at TSA_7 in all sites means that observations in spring 2012 move close back to observations in spring 2011 as normal in terms of angle. Meanwhile, there is some degree of a drop in the angle in TSA_{14}. This angle, consisting of three observations (autumn 2014–spring 2015–autumn 2015), indicates the direction of change in diatom assemblages when the hydrological drought was severe in the catchment. The reduction of the angle is more obvious in sites with low stream order than in sites with stream order 9. This means that diatom assemblages in 2015 autumn did not return close to those in 2014 autumn but unusually moved away from them.

Second, the Survey-based Segment Angle (SSA_{y}) was calculated to see the timing of changes in the angle over time. SSA_{y} is defined as the angle measured between two survey-based segments (SS) made of three consecutive observations in NMDS space and was obtained by the equation 5.7 below.

$$SSA_{\mathrm{y}}Spr/Aut = SS_{\mathrm{n\text{-}2}}\angle SS_{\mathrm{n}} \tag{5.7}$$

where y is year of survey when n is stage and $5 \leq n \leq 20$.

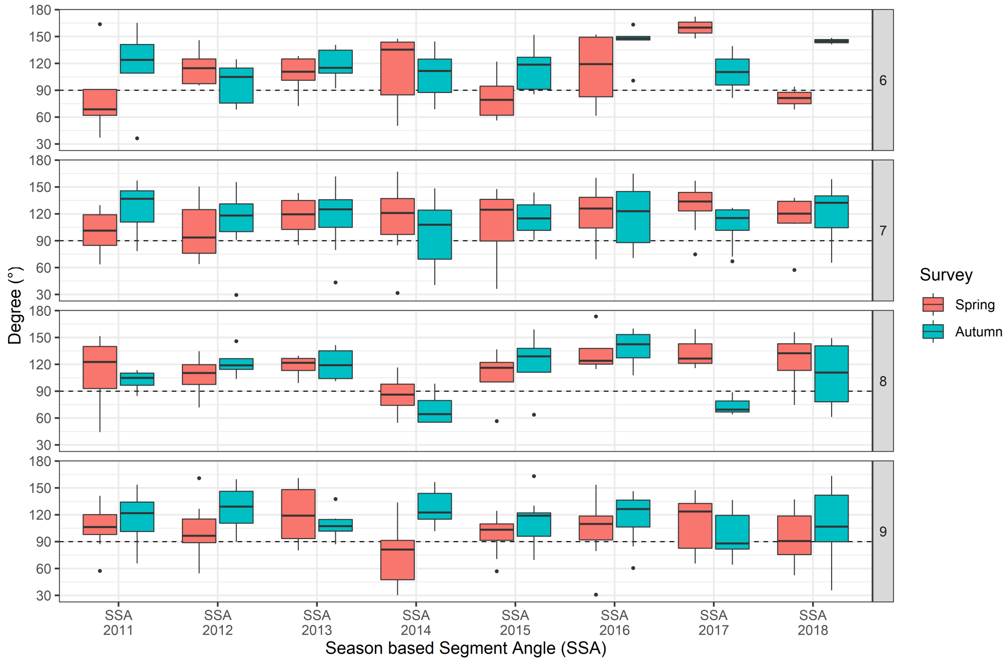


Figure 5.29: Distribution of Survey-based Segment Angle (SSA_{y}) by stream order. Horizontal dashed lines indicate right angle.

Figure 5.29 shows the distribution of Survey-based Segment Angle (SSA_{y}) over time for spring and autumn by stream order. Similar to the results of TSA_{n}, the most frequent angles are obtuse at around 100–120°, while acute angles are rare. Sites with stream order 6 have more variation over time but this cannot be exaggeratedly interpreted because the number of the observations for stream order 6 is limited.

Overall, the distribution of SSA_{y} is steady at obtuse angles without a consistent difference by survey, but there are a few occasions when SSA_{y} sees a significant change. One of them is a big drop in the angle at SSA_{2014} in sites with stream order 8–9. SSA_{2014} is an angle formed between three observations during 2012–2014 in both surveys. A drop in the angle is more significant during the spring survey than during the autumn survey in sites with stream order 9. This means that observations in spring 2014 do not make a turn from the direction to get back to those in spring 2012, but move away from them. This movement corresponds to the straight line in the movement observed in the survey-based trajectories for spring (Figure D.6). However, during the same time, a drop in the angle for autumn is not obvious in sites with stream order 9.

5.3.4.3 Summary of the results of Trajectory Length and Angle

This section summarises the results of Trajectory Length and Angle for the GOVD dataset and links them in an attempt to understand the responses of diatom assemblages to environmental changes such as the construction of the barrages and the hydrological drought during 2009–2018. It is essential to consider both results together as the change of vectors in NMDS space cannot be wholly interpreted without either length (change) or angle (direction).

Unlike the hypothesis that change in diatom assemblages between two surveys would be consistently similar through seasonal changes, thus having a similar length during the whole period, the results of the mean change of Time-series Segment Length (TSL_{n}) demonstrate that changes in diatom assemblage are not consistently similar but different with a constant seasonal change during the period (Figure 5.25). The results of Time-series Segment Angle also demonstrate that observations tend to come back close to where it was a year ago by making a turn at angle 120–160° due to seasonality (Figure 5.28). The best example for displaying similar seasonal changes at similar angles over time is T11a in tributary MR (Figure 5.22), but this is extremely rare. Instead, most sites have different changes over time in terms of length and angle in response to environmental changes.

With regard to the construction of the barrages, the increase of the length at TSL_7–TSL_{10} is the direct response of diatom assemblages to the construction as these are coincidental with the construction period and observed in sites with stream order 9. Despite the disturbance, there is no significant change in trajectory angle. During the same period, TSA_7–TSA_{10} have usual obtuse angles. These results mean that during the construction, the disturbance would have caused more changes to diatom assemblages as suggested by the long length in TSL, but did not affect their seasonality. These interpretations are matched with the location of the observations at stages 3–7 in NMDS space. They are more broadly dispersed than those at stages 11–20 in MNDS space (e.g. M5–M7 in Figure 5.21). As soon as the construction was completed in April 2012, sites with stream order 9 gradually decreased in the length from TSL_{11}, which is supported by the observations during stages 11–18 in NMDS space. They are clustered in close proximity in NMDS space (Figure 5.21). After the completion of the construction in the main river, TSA between observations remained at an obtuse angle, but TSL decreased greatly.

The results of Trajectory Analysis also show the response of diatom assemblages to the hydrological drought that occurred in 2015 in the north region of the Nakdong catchment (Figure 3.5 and 3.6 in Chapter 3). In all sites, there is the least change recorded in the Time-series Segment Length at TSL_{14} and the decrease in the Time-series Segment Angle at TSA_{14}. These responses suggest that observations in autumn 2015 (stage 14) had not returned to where they were in autumn 2014, but ended up being close to observations in spring 2015. These interpretations are well-matched with the location of observations at stages 13–14 in some tributaries (NS, BS, YG, WE, and GM) in the northern areas (Figure 5.22). However, the change at TSA_{14} in sites with stream order 9 is relatively weak in comparison with other sites (Figure 5.28) because of the little impact of the hydrological drought in the main river. Another hydrological drought that struck the study area in 2017 is not identified in the results of Time-series Segment Length (TSL_{18}) and Time-series Segment Angle (TSA_{18}).

5.4 Discussion

5.4.1 How useful are diatom-based indices?

The review of the mTDI results from 2009 to 2018 in the main river and the tributaries demonstrates that river health has been improved to above a satisfactory level since around 2016 across all catchments. This result is contradictory to the worsening trends of water quality variables since around 2012 such as BOD, COD, Chla, and EC, which were demonstrated in Chapter 3 (Figure 3.20 and 3.22). This mismatch appears to be arising from a feature of the index. The TDI index was initially developed based on the sensitivity of species to phosphorus concentration in water (Kelly and Whitton, 1995). The introduction of the mTDI to South Korean rivers was determined by the authority over the DAIpo that applies relationships of diatoms with organic pollution after the assessment that nutrient pollution in Korean rivers was much more severe than organic pollution (Choi, 2006; Noh et al., 2009).

Because of the significant drop of TP concentration in the main river and the tributaries (Figure 3.17) through the period of observations owing to the stricter regulation on phosphorus concentration in water and extensive investment in sewage plants (Figure 3.29 and 3.30), the improvement of river health assessed by the mTDI makes sense. These agreements between the mTDI results and TP concentration in the catchment are in line with results of other studies (Kim et al., 2017; Choi et al., 2019), which were conducted in tributaries NS, and YG, WE and show that the results of the mTDI have high correlations with nutrients such as TN and TP. In addition, in the main river, Lee et al. (2021) found out that the TDI shows positive correlations with TN and PO_4-P, outperforming the DIApo in explaining water quality in the main river. Nevertheless, it is still uncertain why the mTDI score in the study area had such an abrupt rise in 2016 in all catchments, while TP concentration in the rivers was making gradual improvement. Regardless, the review of the mTDI results suggests that the index is an effective tool in terms of monitoring phosphorus concentration in rivers.

The mTDI is considered a robust tool for water quality assessment because it does not respond to changes in the hydrological system. It was revealed in Chapter 3 that during the mTDI monitoring period, there were hydrological droughts in 2015 and 2017, hitting the north and the south regions hard, respectively. Accordingly, diatom assemblages responded to those disturbances by moving differently in NMDS space (observations at stages 13–14 for the hydrological drought in Figure 5.21 and 5.22). However, the mTDI results have no sign of falling or rising at the time of these disturbances. This suggests that the index is a robust means to detect water quality change only, regardless of hydrological changes.

Despite the construction of the barrages having caused a change to the hydrological system (Figure 3.7) and generated disturbances such as the increase and decrease in SS (Figure 3.17), the result of the mTDI appears to have a weak signal. Only six out of the nine catchments in the main river recorded a drop to a very bad state in the environmental grades during 2010–2012 (e.g. ND03–05 and ND07–09 in Figure 5.5). However, the drop to a bad state could not be entirely attributed to the construction of the barrages since the signal is not clear in the overall trends from 2009 to 2015.

To sum up, the use of the mTDI is adequate for assessing water quality or river health in terms of monitoring the response of diatoms to phosphorus concentration in rivers. However, it is not effective at all in assessing other aspects of river health such as organic pollution or environmental changes such as hydrological drought events or construction work in rivers.

Furthermore, the current study demonstrates that the application of the three diversity indices to the GOVD dataset proves to be not very effective in understanding the change in water quality and the environmental changes in the rivers despite some coincidental changes in diversity at the time of disturbances. The results of three diversity indices show that diversity in diatom communities increases in response to the hydrological drought in 2015 (Figure 5.6–5.8). These results are similar to the findings of Boix et al. (2010) that the results of the same three diversity indices to diatom communities are significantly related to hydrological effects in Mediterranean rivers. However, the opposite result also exists as Peszek et al. (2021) find no difference in Species Richness between drought and non-drought periods in Polish rivers.

In relation to water quality, diversity indices produce contradicting results in the literature. It is reported that diversity in diatom communities has a negative relationship with water pollution such as low diversity in polluted water (e.g. Species Richness in a Spanish river, Camargo and Jiménez 2007; Species Richness, Species Evenness, Shannon Diversity Index, Simpson Diversity Index in urban streams in China, Chen et al. 2016). However, other research finds a positive relationship with pollution (e.g. Species Richness in a Chinese river, Tan et al. 2014). This project also finds higher diversity values with the three diversity indices in sites with stream order 9 where water quality is exacerbating since the completion of the barrages in 2012 (e.g. BOD, COD, and Chla). These contradicting results in relation to water quality in the literature and the results of this project support the findings of Blanco et al. (2012) in France. Blanco et al. (2012) reveal that diversity indices with epilithic diatoms (Species Richness and Shannon Diversity Index) exhibit poor linear correlations with trophic levels and discourage the use of diatom diversity indices in biological assessment in rivers.

In this project, the three diversity indices with the GOVD dataset show no significant response to the construction of the barrages and this may be down to sampling frequency and sampling effort. Diatoms have a relatively shorter lifespan (daily timescales) than invertebrates and fish with ease of sampling and these traits are some of the reasons for diatoms to be widely used as indicators in monitoring water quality changes (Round, 1991; Stevenson et al., 2010). However, diatom sampling at the interval of five to six months in between two survey periods for the GOVD dataset would have allowed enough time for diatoms to recover and evolve unless a physical disturbance was persistently strong or was close to the time of sampling. For example, in a study of daily water quality monitoring at one site in the main river for four months in 2011, Hwang et al. (2013) revealed that the construction of the barrages did not significantly affect water quality except for SS and Turbidity and an increase in the two variables was only observed in the event of rain. This result may suggest that disturbance from the construction may be intermittent. A study of Wu et al. (2016) is worth noting as an example, although their research cannot be directly compared with the current study due to differences in the types of diatoms and geographical setting. Wu et al. (2016) found that planktonic diatoms in rivers can change as quickly as in one day during the wet season and as slow as in over 30 days during the dry season by comparing daily diatom assemblages throughout the year in a German river. They concluded that sampling frequency for diatoms should be increased with increasing catchment wetness.

Though the GOVD dataset has been strictly maintained by a long engaging research team and the National Institute of Environmental Research for quality control, however, there is a chance that the effort and time spent on diatom identification remain subjective to the individual analyst. In the national river health monitoring programme during 2009–2018, there have been around one to five new analysts coming in and out every year and these changes may have made an impact on the dataset. As a result, the GOVD dataset sampled twice a year would not be sufficient for detecting the response of diatom communities to the construction of the barrages. Therefore, in order to capture the response of diatoms to a physical disturbance in rivers, it is recommended to increase sampling frequency depending on research objectives or to strategically take samples before and after the targeted disturbance.

5.4.2 What does variability in multiple samples imply for application of diatoms?

The comparisons of observations from the MYD and ROD datasets in NMDS space (Figure 5.10) demonstrate that the relationships among sites in the ordination space are primarily maintained despite the two datasets being sampled at different times of the year (the MYD dataset in October and the ROD dataset in December). Similarly, the comparisons of two observations sampled three weeks apart in the main river from the MYD dataset (Figure 5.13) show that the overall structure, displayed as longitudinal trajectories in NMDS space, remains similar in shape and length despite the significant movement in NMDS space at some sites (M1–M3 and M15) during the time. Also, considering the observations in the GOVD dataset (Figure 5.21 and 5.22) consistently moving from spring to autumn in NMDS space, it is suggested that epilithic diatoms in rivers respond to seasonal changes on a year time scale, although the current study has a limitation that diatom samples throughout the whole year are not successively examined.

The results of CCA using two observations per site in the MYD dataset (Figure 5.15) affirm that water temperature (WT) is the key environmental factor that explains variance most during the three weeks except for at two sites (M1 and T11b). This result highlights the important role of water temperature on diatom assemblages in rivers. Although Kelly et al. (1995) found no significant influence of season on benthic diatom indices using samples collected at four different times of year in rivers in England and Scotland, temporal aspects of diatoms in rivers are overlooked in the literature. For example, only hydrological conditions in rivers are considered in recommendations for sampling epilithic diatoms (Kelly et al., 1998). However, Passy (2007a) revealed that the environmental data itself account for only 9 % of diatom variance, whereas 51 % is captured by time (7 %), space (16 %), and all of their covariance terms (28 %), and stressed the importance of temporal and spatial variables in utilising diatoms for bio-monitoring in rivers. In addition, Zou et al. (2018) constructed seasonal changes of diatom assemblages in a Chinese lake using monthly collected diatom data for two years and concluded that the seasonal diatom succession was driven by water temperature, not trophic level. This result suggests that there is a seasonality in diatoms that naturally thrive and decrease. Consequently, the demonstration with the MYD dataset supports that there needs to be more effort to look at temporal changes of diatoms for better water quality assessment in a river.

The role of water temperature in the experiment with the MYD dataset suggests that there is a possibility that the GOVD dataset may have included variability triggered by changes in water temperature during the two month survey period. Although, this variability by water temperature may not affect the results of indices as no significant influence of season on benthic diatom indices was found (Kelly et al., 1995), the role of water temperature cannot be fully ruled out when the species composition of diatom assemblages is utilised. Variability in diatom assemblages can be susceptible under the current ‘one visit per site’ sampling practice in the national river health monitoring programme. Within the two month period (May–June and Sep–Oct), air temperature and water temperature can change significantly and their changes will likely be more significant due to the effect of climate change. In two headwater streams in the U.K, Snell et al. (2014) found that diatom communities and the result of the TDI are well-matched with stream discharge conditions over the preceding 18–21 days and TP concentrations over an antecedent period of 7–21 days. Therefore, for a more accurate water quality assessment in the current national programme, it is recommended to make effort to minimise variability caused by seasonality or unusual climate conditions by increasing sampling frequency by up to two visits per site at a minimum during the two month period or reducing the two month period to one month. At the same time, more precise temporal variables need to be explicitly integrated into utilising diatoms in water quality assessment such as exact date (day and month) or temperature.

While water temperature explains the changes in diatom assemblages most during three weeks in the Nakdong river and its tributaries, the impact of water temperature is not universally similar across the sites. In this study, stream order, representing the river's hierarchical system, was expected to act as a key variable to determine different responses of diatom communities as in the RCC (Vannote et al., 1980), however, the difference between sites with stream orders 6–9 is not statistically significant (Figure 5.11 (b)). Instead, the spatial distribution of segment length on the map (Figure 5.11 (c) and 5.12) and the results of CCA (Figure 5.16) demonstrate that river impoundment caused by the barrages makes difference in the response of diatom assemblages to water temperature because of changes to the thermal system in the main river. These findings are similar to the results of recent studies in Poland and the U.K. For example, Krajenbrink et al. (2019) discovered in Northern England that river impoundment has caused the homogenisation in diatom assemblages. Similarly, Peszek et al. (2021) found that diatom communities in large watercourses were most similar to each other regardless of season or drought compared to those in small rivers.

Meanwhile, variability at M1 and T11b in the MYD dataset appears to be related to local issues and these are the examples presenting the importance of understanding local knowledge for an appropriate and better application of diatoms in water quality assessment. Site M1 is the best example in terms of site selection. While the results of CCA show that the second sample in most sites moved to the top in ordination space in response to changes in water temperature, two observations from M1 are positioned on the left side of the space with almost no change to the direction of water temperature (see M1 in Figure 5.15 (b)). This is entirely down to the unique local hydrological system. Site M1 is located in the lower reservoir of the Andong Dam (see Figure 3.28 (b)), which is used for a hydroelectric power plant. As a result, water in the reservoir is mainly stagnant and its physical and chemical variables are different to free-flowing water. For example, during the fieldwork, the DO level at M1 was measured at 3.1 and 7.6 mg/L each, while the other sites ranged from 9.0 to 12.0 mg/L (The measured values for environmental variables were cross-checked with government water monitoring records from Water Environment Information System (2019) and they were almost the same). This suggests that the structure of diatom assemblages and their responses at M1 are likely different to others. Therefore, it may not be ideal to use this site as a diatom sampling point for water quality assessment in comparison with other sites in the rivers. For a better evaluation of water quality, it is recommended to set up a new monitoring point below the reservoir. A significant change in T11b is likely to be the effect of anthropogenic activities. Like M1, the results of CCA show that the second observation at T11 moved along axis 1 with almost no change along axis 2. For instance, T11b recorded fairly similar values in both surveys for WT, DO, and pH, but had a substantial difference for other variables: resistivity (6129 and 5084 Ohms), conductivity (163.1 and 196.6 $\mu S\,cm^{-1}$), TDS (122 and 160 mg/L) and ORP (197 and 238 mV). The decrease in resistivity and an increase in conductivity and TDS indicate the influx of salts, nutrients, and sediments into the water and these changes are thought to be the influence of construction work that was underway on the riverside in tributary MR during the fieldwork.

To summarise, the use of diatoms in water quality assessment primarily stands on the sampling practice with one visit per site at low and stable flow to capture variability in diatom assemblages on the average state. However, the experiment with the MYD dataset demonstrates that even in those riverine conditions, diatom assemblages in rivers can change as quickly as within three weeks and water temperature is the key environmental factor in explaining the change. This result highlights the importance of water temperature on diatom assemblages, and temporal aspects of diatom assemblages in rivers need to be investigated more. This result also suggests that the current sampling method in the national river health monitoring programmes has a chance of capturing variability caused by water temperature rather than pollution-related variables depending on sampling timing. This variability will be more susceptible under the current method and would make it difficult for accurate assessment moving forward because of the effect of climate change. Therefore, in the national programme, the sampling method needs to be changed by increasing sampling frequency by up to two at a minimum during the two month period or reducing the two month period to one to minimise variability in diatom assemblages. Also, temporal variables need to be included in using diatoms for an accurate water quality and river health assessment.

5.4.3 What are the driving forces that lead to changes in diatom assemblages?

The projection of the observations from the GOVD dataset from 2009 to 2018 using NMDS analysis provides pictures of the changes of diatom communities in time-series order by river and site. The results of NMDS analysis are applied to the Community Trajectory Analysis, which serves as the platform to understand the responses of diatom communities by measuring geometric features of the observations (i.e. length and angle) in ordination space.

The results of NMDS analysis for the GOVD dataset (Figure 5.21 and 5.22) show that diatom assemblages across the sites have consistently evolved with a periodic movement through changes of seasons over time. Under stable and undisturbed conditions, the extent of the change is limited, mainly remaining in a similar position of the ordination space (e.g. T8a, T8b, T11a, T11b in Figure 5.22) with a regular shift between spring and autumn. However, this is quite rare in the study area. In most cases, observations in the time-series order have moved around in the space with the seasonal shift from spring to autumn. Those sites are divided into two groups by their movements; the observations in the first group (e.g. M4–M10, M13–M15, T2, T4, T10) have completely relocated themselves to the second quadrant from the first or fourth quadrant in NMDS space over time, while the second group (e.g. M1, M3, M12, T1, T3, T5, T6, T7, T9) have randomly moved around in a similar space after going through abrupt changes.

These contrasting responses across the sites in NMDS space suggest that there are different factors controlling diatom communities in the rivers over time, although the position of observations and the direction of movements in NMDS space do not directly indicate any meaningful interpretation in terms of environmental variables. Particularly, in the main river, significant changes in the shape of longitudinal trajectories as well as the movement of centroids in the space (Figure 5.18 and 5.19) suggest that there are significant environmental changes provided the fact that two longitudinal trajectories of the main river from the MYD dataset remain similar in shape despite the different timing of sampling. Significant and permanent movements of observations in NMDS space are mainly identified in sites from the first group (e.g. M4–M10, M13–M15, T2, T4, T10), which are directly influenced by the barrages. Also, those sites have their observations at stages 10–20 dispersed in close proximity in NMDS space (Figure 5.21). Unfortunately, there have been no studies carried out on the response of diatom assemblages in relation to the barrages, but these movements in NMDS space are similar to the results of other studies (Wu et al., 2009; Krajenbrink et al., 2019) which look at the impact of dams on diatom communities. For example, Wu et al. (2009) found that diatom observations made a movement and created clusters in NMDS space between upstream and downstream sites after the dam construction in a tributary of the Xiangxi River in China. Similarly, Krajenbrink et al. (2019) revealed that diatom observations from control sites to dams were more clustered in NMDS space compared to observations in downstream sites in Northern England because of river impoundment.

The breakdown of the trajectories made of the observations from the GOVD dataset into geometric features (segment length and angle) provides a clear picture of different degrees of changes in diatom assemblages during 2009–2018. The results of trajectory length and angle show that changes in diatom assemblages in the main river were the biggest in spring 2012 right after the completion of the barrages in April 2012 and the changes remained significant until spring 2014 when they started to diminish. For instance, the results of Time-series Segment Length (*TSL*) by stream order (Figure 5.24 and 5.25) show that sites with stream order 9 had a peak at TSL_7 and remained high until TSL_{11}, which is spring 2014. Afterwards, TSL gradually started to decrease. This means that even after the completion of the barrages, diatom assemblages in those sites changed significantly along with a seasonal oscillation during 2012–2014. These changes are also identified in the results of Survey-based Segment Length (SSL). Figure 5.26 shows that SSL in sites with stream order 9 started to decrease at SSL_{2014}Spr and SSL_{2014}Aut. In trajectory angle, the results of Time-series Segment Angle (TSA) do not show any notable change over time but the results of Survey-based Segment Angle (SSA) show the dominance of acute angles at SSA_{2014}Spr. This means that observations in spring 2014 continued to move away in the direction of the previous two spring observations between 2012–2013. This acute angle is well-matched with a linear movement of three observations in NMDS space (stages 7–11 in Figure D.6), but the change to acute angle is not obvious in autumn.

Since 2014, the movement of diatom observations in NMDS space appears to dwindle significantly and the observations remain in a similar position with little seasonal change. This result means that it took two years after the completion of the barrages until diatom assemblages in sites affected by the barrages are completely changed. This result is in line with the findings of Wu et al. (2009). In their study, Wu et al. (2009, Fig.5) presented the distribution of diatom observations in NMDS space using samples collected before, one year, two years, and three years after the dam construction in a Chinese river. Their results show that diatom observations were scattered in NMDS space without a group one year after the dam construction and they made clusters in ordination space two years after the construction. Since 2014, little movement in NMDS space at those sites means a short length at TSL and SSL. This result indicates the effect of river impoundment and reaffirms the modification to the thermal system in the main river which is demonstrated in the experiment with the MYD dataset (Figure 5.15) showing the least response of diatom assemblages to changes in water temperature in the regulated reaches of the main river during the three weeks. The effect of river impoundment on diatom assemblages is also supported by the results of Krajenbrink et al. (2019) that revealed the homogenisation of diatom assemblages in impounded sites to dams in Northern England.

Ultimately, the movement of observations in NMDS space and the results of trajectory length and trajectory angle at sites from the first group are evidence of the impact of the barrages on the riverine ecosystems. These results also highlight the decisive role of flow regime in determining aquatic communities in riverine environments (Bunn and Arthington, 2002). Likewise, T2 in the tributaries has a time-series trajectory that ended up on the other side of the space during the period and this may be explained by the operation of the Yeongju Dam (D_YENJ in Figure 3.2), which was completed in the summer of 2016. It is reported that the operation of the dam caused dramatic changes in the river channel of tributary NS such as the regulated discharge, the decrease of sediment supply, and the vegetation succession (Lee et al., 2019a; Lee et al., 2019b).

The results of TSL and SSL by stream order are also effective in detecting the responses of diatom assemblages to the hydrological drought in 2015 that are characterised by the unique movements at stages 13–14 in the time-series trajectory in NMDS space (Figure 5.22). The results of Time-series Segment Length (TSL) by stream order (Figure 5.24 and 5.25) show that TSL significantly dwindled at TSL_{14} in all sites. This means that observations in autumn 2015 were not significantly different to those in spring 2015. In contrast, the results of Survey-based Segment Length (SSL) by stream order (Figure 5.26) show that SSL significantly increased at SSL_{2015}Spr and SSL_{2015}Aut. This means that observations in 2015 were significantly different to observations in the previous non-dry year. To wrap up the response of diatom assemblages to the hydrological drought in 2015, diatom assemblages in 2015 are different to diatom assemblages in the non-dry years and diatom assemblages taken in the spring and autumn of 2015 are similar. Moreover, these phenomena are more obvious in sites with stream orders 6–8 than in sites with stream order 9. These findings in relation to the drought are analogous to the findings of two research. Boix et al. (2010) found the clustering of diatom observations in NMDS space using four samples (April, May, July and September) throughout the year during a strong hydrological drought in Spain. This result means that diatom assemblages during the drought are similar to each other regardless of the season. Peszek et al. (2021) also found that changes in diatom assemblages during the drought period were much smaller in large watercourses than in small rivers.

In NMDS space, sites (e.g. M1, M3, M12, T1, T3, T5, T6, T7, T9) from the second group have moved around and come back to where they were after going through abrupt changes. This movement at those sites appears to be related to other construction work that took place at each site at various timings; weir or bridge construction (Figure 3.28), river embankment maintenance and riverside development. These construction works are insufficient to alter the entire flow regime permanently but sufficient to temporarily disturb the flow at a local scale. In these cases, once the disturbance is over, diatom assemblages then tend to come back to the previous location in ordination space. As shown in the water quality analysis, M1 was affected by the bridge construction, while M2 and M3 were located downstream of the weir construction. In particular, the movement of observations at M2 and M3 is important as their movements in NMDS space are different to those at sites (e.g. M4–M15) from the first group affected by the barrages (Figure 5.21). These examples demonstrate different impacts of weirs and barrages on flow regime and diatom assemblages. In the tributaries, T3–T6, T9, and T11b were affected by nearby construction that include river embankment maintenance and the building of a new bridge.

The present study explains the responses of diatom assemblages over time in relation to the flow regime such as the construction of the barrages and the hydrological drought at the watershed scale. Nevertheless, at the same time, other variables such as land use, riparian vegetation, supply of sediments, and biotic factors would have contributed to the change in diatom assemblages. However, their impacts may be relatively less considerable than climatic variables or flow regimes at this watershed scale (Stevenson, 1997).

5.5 Conclusions

This chapter has dealt with the GOVD and MYD datasets in an attempt to answer research questions 3–5. First, the results of the mTDI and the performance of diversity indices on the GOVD dataset were evaluated to see their effectiveness in understanding changes in water quality and riverine environments. Then, using the MYD dataset, established on fieldwork, variability in diatom assemblages in multiple samples in autumn was examined to find out the setbacks in the current GOVD sampling period as well as to find a better way of using diatoms in the future. Lastly, as a holistic approach to assessing river health, observations of diatom assemblages in the GOVD dataset over time were analysed using NMDS analysis and CTA to understand their responses to changes in riverine environments such as the construction of the barrages. Key findings from this chapter are:

1. The review of the mTDI result indicates the significant improvement of river health from 2009 to 2018 in all catchments in the main river and the tributaries, achieving a satisfactory level or beyond from a bad state or a very bad states of river health in the environmental grades. This improvement at the catchment level is in accordance with a gradual decrease in phosphorus level but in disagreement with the worsening trends of water quality such as BOD, COD, and Chla which are demonstrated in water quality analysis. The results of the mTDI do not show any sign of change in the trends in connection with the construction of the barrages or the hydrological drought. Therefore, the index is a good and robust tool for monitoring river health in terms of phosphorus concentration in water. However, it is not beneficial to use the index in an attempt to assess different facets of water quality or understand comprehensive river health in relation to changes in riverine environments.

2. The application of three diversity indices — Species Richness, Species Evenness and Shannon Diversity Index — to the GOVD dataset demonstrates that temporal changes of diversity in the three diversity indices from 2009 to 2018 provide a limited explanation of river health regarding the changes in water quality, the construction of the barrages, and the hydrological drought. The results show that diversity in the three indices all increased in the hydrological drought in 2015. Regarding water quality, Species Richness increased in sites with stream order 9 where water quality is worsening with an increase in BOD, COD, and Chla since the completion of the barrages in April 2012. This result means that the indices have a positive relationship with water pollution. Diversity in the three indices shows no significant change in relation to the construction of the barrages and this may be down to sampling frequency and sampling effort. The GOVD dataset has observations sampled twice a year at an interval of five to six months and this gap would have allowed enough time for diatoms to recover and evolve unless the disturbance from the construction was consistent or close to the time of the sampling. In addition, even though the GOVD dataset has been maintained by the research team and the authority for quality control, nevertheless, new analysts coming in and out every year might affect the sampling and identification effort. Therefore, it is difficult to examine the impact of the barrages using indices and there needs to be a change in sampling frequency or timing to detect the responses of diatoms to physical disturbances such as an increase in sampling frequency or a sampling before and after a targeted disturbance.

3. The projection of observations from the MYD and ROD datasets into NMDS space demonstrates that despite the two datasets sampled at different times of year (autumn and winter in 2018), observations from the same sites are located in a similar position, maintaining their relationships to each other in the space. This suggests that diatom assemblages across the sites similarly respond to seasonal changes. The comparison of segment length between

observations in the MYD and ROD samples shows that the distance between the MYD samples is larger than the distance within the groups in the ROD dataset but shorter than the distance between the groups in the ROD dataset. This result means that changes in diatom assemblages during the three weeks are larger than those among the samples taken at the same time within the groups but smaller than those among the samples between the groups. During the period, the changes in diatom assemblages in the MYD dataset are not significant by river type or stream order. The visualisation of the changes on the map shows that little change in diatom assemblages is made in sites regulated by the barrages, while big changes are observed in the tributaries or the up-, and downstream of the main river where flow is free running.

4. Two observations in the MYD dataset collected from the main river are subsetted and compared as geometric vectors (trajectory shape and length, and segment length). Despite the significant changes in the upstream and downstream sites (M1–M3, and M15) during the three weeks, the two longitudinal trajectories are similar in shape and length. The breakdown of the trajectories into segments displays a similar distribution of segment length between the two surveys. The ECDF and the K–S test confirm that two sets of segment length distributions are likely from the same distribution. Therefore, despite some changes in diatom assemblages between the two surveys, relationships between sites from a longitudinal perspective remain similar by responding to environmental changes.

5. The results of CCA for observations in the MYD dataset show that water temperature is the key environmental factor in explaining variability during the three weeks except for at two sites (M1 and T11b). This result highlights the importance of water temperature on diatom assemblages in rivers and there needs to be more effort to look at temporal changes in diatom assemblages. In addition, it also means that the current practice of 'one visit per site' for diatom sampling within the two month period in the national

river health monitoring programme is susceptible to variability in diatom assemblages. Therefore, in the national programme, the sampling method needs to be changed by increasing frequency by up to two per site at a minimum during the two months or reducing the two months to one to minimise variability. At the same time, temporal variables need to be more integrated into using diatoms for accurate water quality and river health assessment. The results of CCA by stream order show that during the three week period, changes in diatom assemblages are small in sites with stream order 9 compared to other sites with stream orders 6–8. This evidence suggests that sites with stream order 9 have a different thermal system due to the impact of the barrages.

6. As a holistic approach to assessing river health assessment, the results of NMDS analysis using the GOVD dataset show that observations have constantly moved in NMDS space because of various factors such as seasonality, hydrological change and anthropogenic activities in the rivers. Under undisturbed conditions, diatom assemblages in NMDS space have remained in similar positions with only seasonal changes but this is rare. In contrast, most sites have gone through significant changes in the space over time and are divided into two groups by movement. The first group has mainly sites in the main river affected by the barrages and their observations have moved to the other side of the NMDS space and reestablished. Sites in the second group have observations that have returned to their previous location in the space after going through abrupt changes. The reasons for different movements in NMDS space are that the barrages have completely altered the flow regime in the main river, resulting in significant changes in diatom assemblages in sites from the first group. Sites in the second group were temporarily affected by other construction work that are not sufficient to change the hydrological system but to generate disturbance in flow temporarily.

7. The application of Community Trajectory Analysis to the results of NMDS

analysis provides a whole picture of changes in diatom assemblages from 2009 to 2018 as geometric vectors (length and angle). The distribution of the Time-series Segment Length (TSL) by stream order shows that sites with stream order 9 reach the highest in length at TSL7 and remain high before starting to decrease from TSL11. This means that changes in diatom assemblages during 2012–2014 were significant compared to the pre-construction period. These changes are also identified in the results of the Survey-based Segment Length (SSL). Since 2014, TSL in those sites started to decrease from TSL11 and remain short. This result means little difference in diatom assemblages between spring and autumn and evidence of the modification to the thermal system in the main river due to river impoundment by the barrages. This result is in agreement with the results of the MYD dataset. The results of TSL and SSL show the response of diatom assemblages to the hydrological drought in 2015. The shortest length at TSL14 and the long length at SSL2015 mean that observations in spring 2015 are very similar to those in autumn 2015, while observations from spring and autumn in 2015 are different to those in the previous non-dry year. These changes are more significant in sites with stream order 6–8 than in sites with stream order 9.

8. The distribution of the Time-series Segment Angle (TSA) shows that obtuse angles are common between observations in a time-series order regardless of stream order. This means that diatom observations in all sites tend to move back close to where they were a year ago due to seasonality and there is no obvious change in TSA over time. The distribution of the Survey-based Segment Angle (SSA) by stream order shows that sites with stream orders 8–9 have a significant drop to the right or acute angles in 2014 (SSA2014) and this phenomenon is more obvious in spring than in autumn. This means that diatom observations in spring 2014 tend to move straight away from their average position in NMDS space, and move into the other side of NMDS space. After this movement, diatom observations tend to stay in a similar

position of the space.

5.6 Summary

This chapter has looked at the effectiveness of diatom-based indices in explaining river health in relation to changes in water quality, and riverine environments in the Nakdong catchment. The results show that they are not effective in detecting the impacts of the barrages on diatom assemblages despite some strengths. This chapter has also examined variability in multiple diatom samples during the GOVD autumn sampling period. The results demonstrate that water temperature is the key factor and there needs to be more effort to investigate diatoms in temporal aspects. Lastly, it has examined the responses of diatom assemblages in the GOVD dataset using NMDS analysis and Community Trajectory Analysis (CTA). The results of trajectory length and angle demonstrate the different extent of changes in diatom assemblages in relation to the construction of the barrages and the hydrological drought. The application of CTA is effective in comparing and detecting changes in diatom assemblages based on geometric vectors such as length and angle.

In the next chapter, I will combine the results of water quality analysis from Chapter 3 with the responses of diatom assemblages in the GOVD dataset from Chapter 5 and will examine their spatio-temporal changes based on relationships between the two. Ultimately, these analyses will help understand river health in the Nakdong catchment from 2009 to 2018 based on relationships between water quality and diatom assemblages.

Chapter 6

Relationships between environmental variables and diatoms

6.1 Introduction

This chapter addresses research question 6 (section 2.6.2) by linking the results of water quality analyses in Chapter 3 with those of diatom analyses in Chapter 5 in order to holistically understand river health in the Nakdong river and its tributaries from 2009 to 2018 based on relationships between water quality and diatoms.

There are three goals to be accomplished. First, the GOVD dataset will be divided into groups based on similarity and their spatio-temporal pattern will be examined to see how they have evolved in time and space. Second, the obtained diatom groups will be characterised by summarising water quality as well as finding out one or more indicator species per group. This step determines the feature of diatom groups by linking water quality and indicator species and helps interpret the meaning of the spatio-temporal pattern of diatom groups in the catchment. Last, Canonical Correspondence Analysis (CCA) will be performed to find out relationships between environmental variables and observations. In addition, these results will be further inspected in combination with stream order and stages of construction for the barrages to understand more details on how the relationships have changed in time and space. Consequently, these analyses help achieve a holistic understanding of river health in the Nakdong river and its tributaries in relation to the construction of the barrages. They also provide an insight into what variables/species can be used better for river health assessment moving forward in the Nakdong river and its tributaries.

6.2 Methods and datasets

To complete the tasks above, methods, datasets, and expected outcomes are presented in the flow chart (Figure 5.1). More detailed methods are explained below.

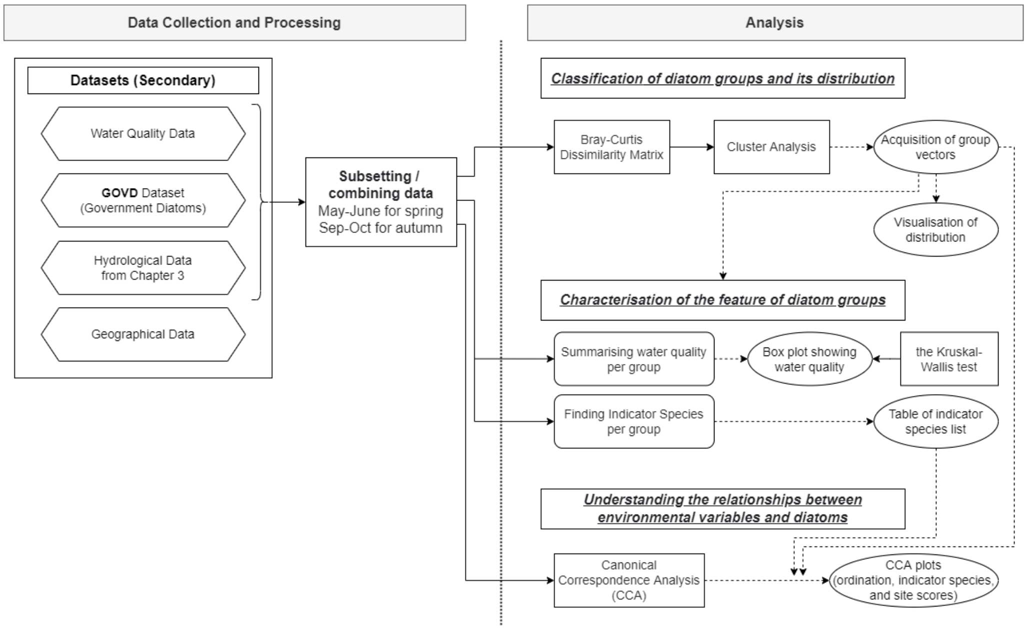


Figure 6.1: Data requirements and workflow for a comprehensive understanding of river health in the Nakdong catchment

6.2.1 Classification of diatom groups and its distribution

The first goal of this chapter is to create the spatio-temporal pattern of diatom groups using the GOVD dataset and understand their patterns in time and space. Accordingly, cluster analysis was performed with the same GOVD dataset from Chapter 5 that includes all observations taken in both surveys (spring and autumn) from 2009 to 2018 in the main river and the tributaries.

As the first step to analyse the dataset, a dissimilarity matrix was calculated using the Bray–Curtis dissimilarity method implemented by the vegdist function in the vegan package (Oksanen et al., 2019). The obtained distance matrix was hierarchically clustered by the hclust function with the Ward's minimum variance method to visually create a dendrogram using the factoextra package (Kassambara and Mundt, 2020). The number of groups was manually determined based on their distances in the dendrogram. The obtained diatom groups then were visually displayed in the matrix form as well as on the map to see the distribution of diatom groups in time and space.

6.2.2 Characterisation of the feature of diatom groups

The next task is to define the characteristics of the obtained diatom groups by water quality and diatoms for a better understanding of their distributions. Thus, each group was characterised by the statistical summary of water quality and discharge, and the identification of indicator species per group.

With regards to defining the feature of water quality per group, water quality data measured during the two diatom surveys (May–June and September–October) were subsetted from the whole water quality data used in Chapter 3. Because diatom samples were only collected once per site during each survey period, the subset water quality data were averaged by each survey to ensure that the average state of water quality is captured and matched to the corresponding diatom observation. Water quality parameters are water temperature (WT, ℃), pH, dissolved oxygen (DO, mg/L), biological oxygen demands (BOD, mg/L), chemical oxygen demand (COD, mg/L), total organic carbon (TOC, mg/L), suspended solids (SS, mg/L), electrical conductivity (EC, µmhos/cm), chlorophyll-a (Chla, mg/L), total nitrogen (TN, mg/L), NH_3N, and total phosphorus (TP, mg/L). In addition, discharge at each site during the survey periods was processed in the same way. In the case where discharge data were unavailable at sites, they were acquired or calculated from other gauging stations or dams nearby (see the locations in Figure 3.2). For example, flow records at M1 were borrowed from the Andong Dam (D_ANDO). Flow data at T1 were obtained by the sum of discharge from the Imha Dam (D_IMHA) and flow station (BB3) in the adjoining stream. Flow data at M2 were obtained by the sum of M1 and T1, which meet together to flow into M2 in the main river. The obtained discharge data were then paired with the water quality data and the obtained group vectors. The combined data were summarised in the box plot by the groups before group means for each variable were statistically compared using the Kruskal–Wallis test.

The attributes of the diatom groups were also understood with the identification of indicator species per group. Accordingly, the GOVD dataset, paired with the group vectors, were used for Indicator value analysis. The analysis, performed using the multipatt function with the individual-based method in the indicspecies package (De Cáceres and Legendre, 2009), returned the list of diatom species per group with statistical significance (*stat*) based on the probability of component A and B. Afterwards, each species in the list was assessed for its possibility to be considered as an indicator species to represent its group in consideration of statistical significance and two components. Component A (specificity or positive predictive value) is the sample estimate of the probability that the surveyed site belongs to the target site group given the fact that the species has been found, while component B (fidelity or sensitivity) is the sample estimate of the probability of finding the species in sites belonging to the site group (De Cáceres and Legendre, 2009). For instance, if a certain species has 1 for component A, it means that all sites that contain the species in the dataset belong to the targeted group. If it has 1 for component B, it means that all sites belonging to the targeted group include this species. From a theoretical perspective, it is best to choose taxa as indicator species whose values are as close as to 1 as possible. However, the GOVD dataset, collected during spring and autumn throughout the year, may include taxa that seasonally thrive because of seasonality. As a result, a median of 0.5 was selected as a criterion for *stat* in consideration of components A and B each.

Finally, the results from the summary of water quality and discharge, and indicator species were assembled to define the feature of the obtained diatom groups. Ultimately, these analyses help interpret what the distribution of diatom groups in time and space means in connection with the construction of the barrages.

6.2.3 Understanding of relationships between environmental variables and diatoms

The final task of this chapter is to understand the relationships between environmental variables and diatoms and how their relationships have changed during

2009–2018 in the Nakdong river and its tributaries. Thus, Canonical Correspondence Analysis (CCA) was carried out using the combined datasets above. The analysis was performed using the cca function in the vegan package (Oksanen et al., 2019). The explanatory variables are 12 water quality parameters and discharge. In data processing, observations with NA in any water quality variables were removed. The number of observations used for CCA was 412 out of 613. The results of CCA were interpreted in connection with various variables such as the construction stage for the barrages (Pre-Construction, During-Construction, and Post-Construction) and stream order.

6.3 Results

6.3.1 Distribution of diatom groups in time and space

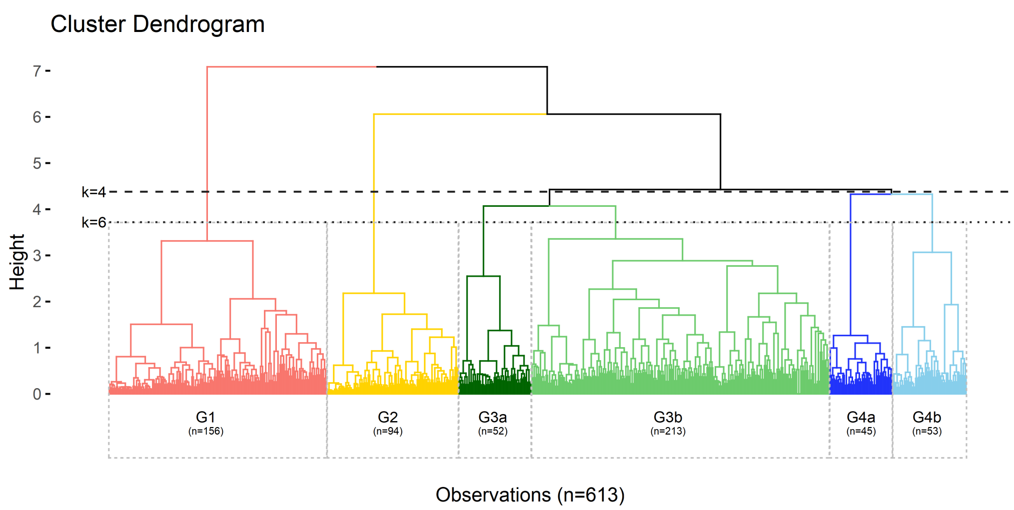


Figure 6.2: Dendrogram of cluster analysis for the GOVD dataset. The GOVD dataset was divided into four main groups ($k = 4$) with two subdivisions ($k = 6$).

Figure 6.2 shows that observations in the GOVD dataset can be divided into four main groups — G1, G2, G3 and G4 ($k = 4$) — with the latter two having two subdivisions a and b each ($k = 6$). In terms of distance, groups G3 and G4 are the closest to one another, and the furthest to G1 following G2. In addition, G3 is the most dominant diatom group ($n = 265$) during the ten years from 2009 to 2018 in the Nakdong river and its tributaries followed by G1 ($n = 158$), whilst G2 and G4 have 94 and 98 observations each.

Each observation, paired with the classified group value, was visualised in the matrix form as well as on the map to see the distribution of diatom groups in time and space. Figure 6.3 shows that the distribution of diatom groups substantially differs in time and space in the rivers. For instance, until 2010 in the main river, there was a clear division of diatom groups between M8 and M9 where the lower reaches of the main rivers are dominated by G3b in comparison to the dominance of G1, G2 and G4b in the upper reaches of the main river. This division point in space coincides with the confluence of tributary KH from Daegu to the main river where pollution levels reach the peak in the main river (see M10–M12 in Figure 3.25) due to the significant contribution of pollution from tributary KH (see T7a and T7b in Figure 3.27). From autumn 2010, G2 in the upper reaches was increasingly replaced with G3b and the whole main river then turned to a dominant state of G1, G3a, and G3b.

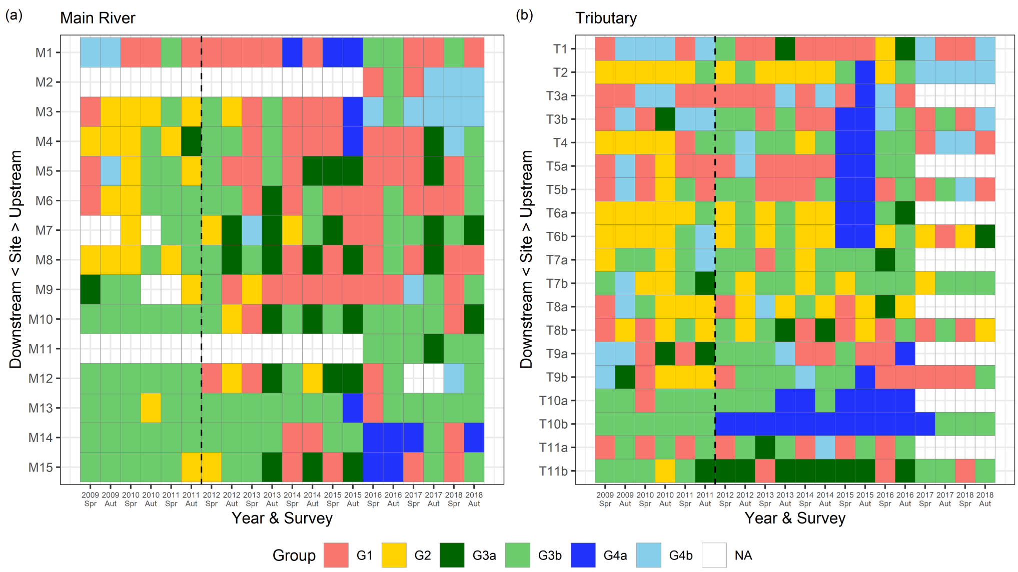


Figure 6.3: Matrix showing the spatio-temporal distribution of diatom groups ($k = 6$) in the GOVD dataset. (a) The main river, (b) The tributaries. The dashed vertical line indicates the time when the construction of the barrages was completed in the main river, while empty cells indicate no data.

Moving onto the distribution in the tributaries, it is significantly different by rivers.

Most rivers (e.g. T1–T6, and T8–T9) are primarily defined by the alternating occurrence of G1, G2, or G4b between spring and autumn, while T7 (Tributary KH), T10 (Tributary NG), and T11 (Tributary MR) have mostly G3b in dominance. These three tributaries are in common in terms of flowing from major cities such as Daegu, Jinju, and Miryang (see their locations in Figure 1.1). In 2011, the emergence of G3b happened in some tributaries as in the main river, but their signal was not obvious. For example, some sites (e.g. T3b, T4, T5b and T6) had intermittent occurrences of G3b since 2011.

One of the notable features in the distribution of diatom groups is that the appearance of group G4 is limited in time and space in the rivers, unlike other diatom groups. The occurrence and dominance of subgroup G4b are geographically confined in the most upstream sites (e.g. M1–M3) in the main river or some sites (e.g. T1 and T2) in the tributaries. Those sites are generally located in the north region of the study area. In contrast, the development of G4a is more distinct in the tributaries than in the main river. G4a is primarily found in the most upstream and downstream sites of the main river (e.g. M1–M4, M13–M15) on a few occasions including 2015 autumn, while the tributaries have a dominance of G4a in sites (e.g. T2–M6b) in the north region for the whole of 2015 and with a constant dominance in T10 (e.g. T10a and T10b) from 2012 to 2017. This means that subgroup G4a is more dominant and widespread in the tributaries than in the main river, lasting throughout the entire year of 2015 in comparison to the main river.

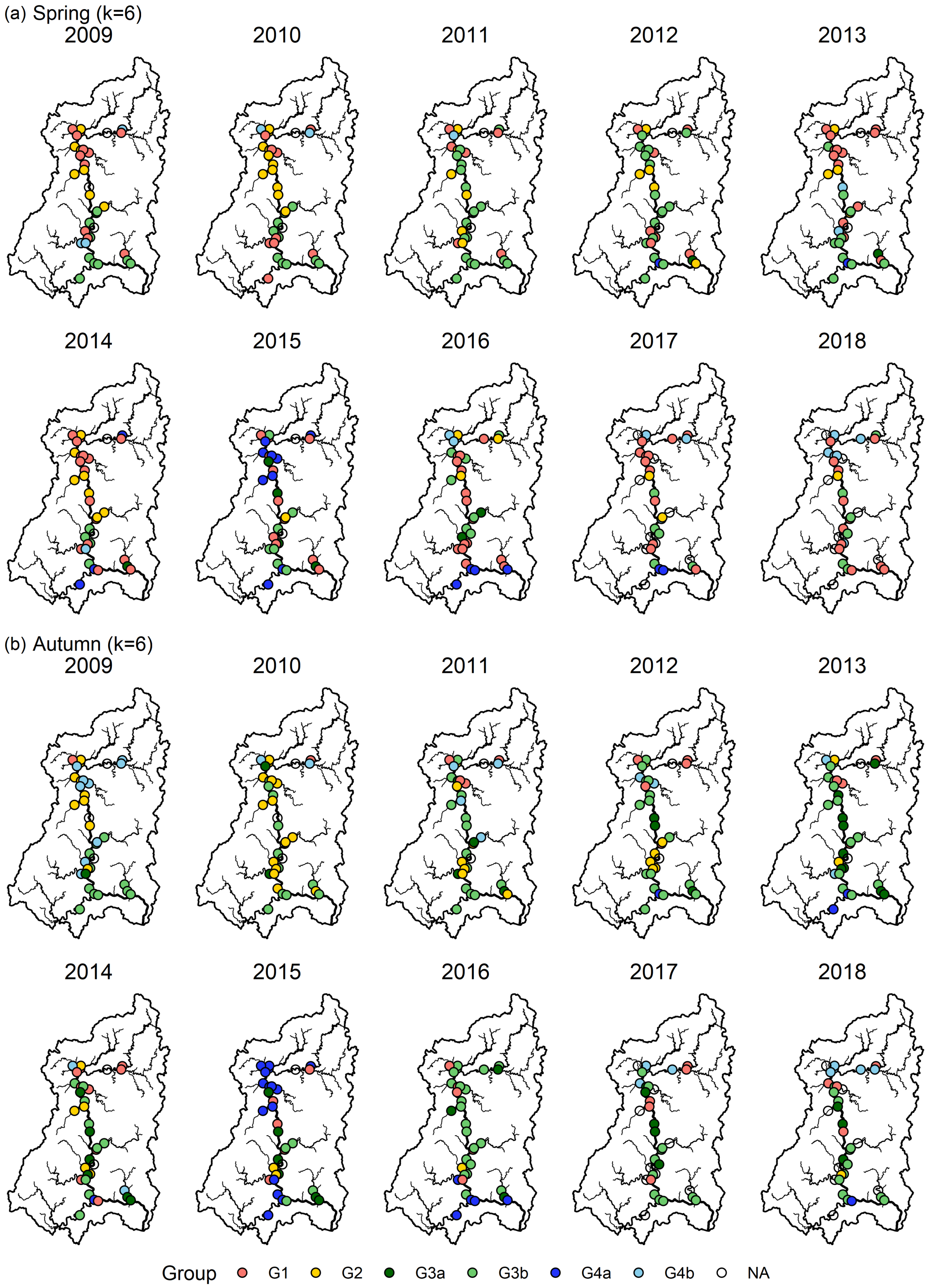


Figure 6.4: Map showing the spatio-temporal distribution of diatom groups in the GOVD dataset. (a) Spring ($k = 6$), (b) Autumn ($k = 6$). Open circles (NA) indicate no data.

6.3.2 The features of diatom groups

In this section, the classified diatom groups were characterised by the statistical summary of water quality and the identification of indicator species. Consequently, these results help understand what the distribution of the diatom groups in time and space means based on the features defined by water quality and indicator species per group.

6.3.2.1 Characteristics of water quality for diatom groups

The statistical summary of 12 types of water quality variables and discharge during the two survey periods are presented in Figure 6.5 by diatom groups. The classified water quality for the groups was examined using the Kruskal–Wallis test to compare the difference in group mean. The results of the test show that diatom groups have a significantly different mean for all variables except WT. The features of water quality are discussed with their frequencies by stream order (Figure 6.6).

The first diatom group G1 is defined with a moderate range of values in BOD and COD, and a low level of nutrient pollution compared to other groups. For example, it has a low concentration in Chla, TN, and TP, despite a high concentration of NH_3N. This group is also marked with low discharge and low SS. This group appears in all stream order sites but is most commonly found in sites with stream order 7.

Group G2, more commonly found in low stream order sites, is characterised by a high concentration of nutrients and a low level of organic matter. For instance, compared to G1, this group is considerably much higher in TN and TP concentrations, but lower in BOD and COD. Another feature of this group is that despite the high concentration of nutrient pollution, the Chla level is lower than in G1 where nutrient concentrations (TN and TP) are the lowest in the groups. In terms of discharge, this group is similar to G1. It shows a relatively higher level of SS than G1, but a lower level of SS than G3b.

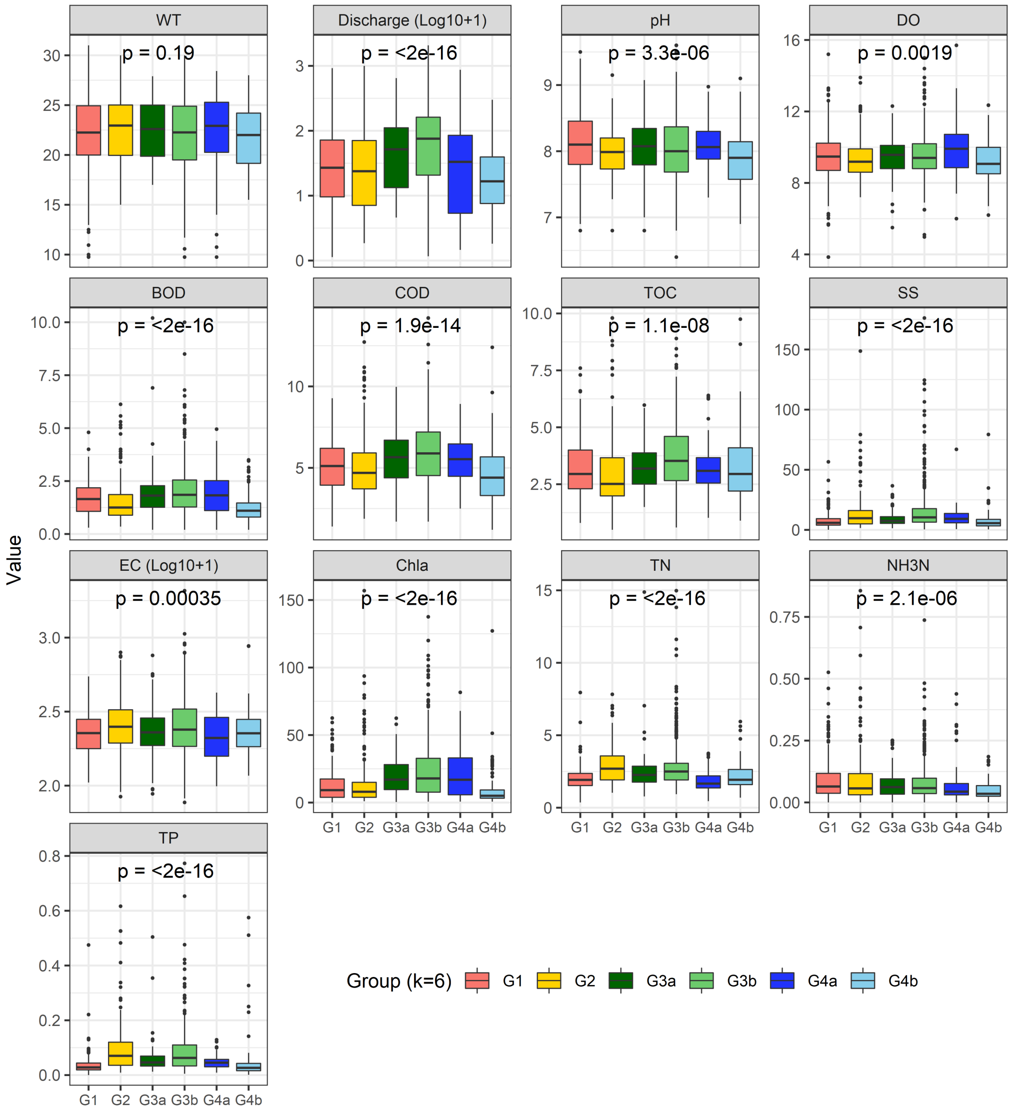


Figure 6.5: Box plot showing the statistical summary of water quality and discharge during the two survey periods by diatom groups ($k = 6$). The results of mean comparison by diatom groups using the Kruskal–Wallis test are shown on each plot as p.

Meanwhile, group G3 is described by the highest level of organic (BOD and COD) and nutrient (TN and TP) pollution as well as a high concentration of Chla with the biggest discharge among the groups. This group is typically found in sites with stream order 9, accounting for more than half of the total number of sites, so the main river is strongly dominated by the occurrence of a cluster of G3. Within G3, subgroup G3b shows higher pollution levels than G3a across all organic and

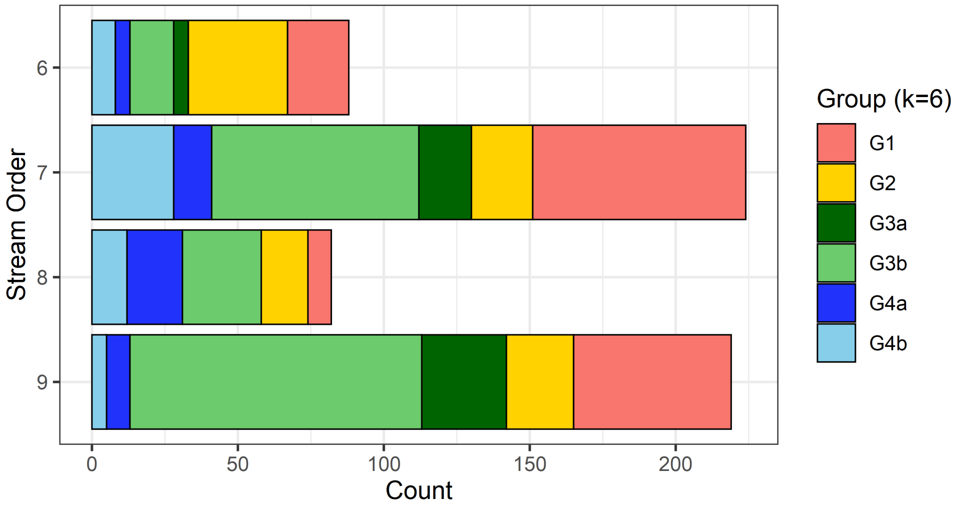


Figure 6.6: Bar plot showing the frequency of diatom groups ($k = 6$) by stream order

nutrient variables. For example, G3b has a high level of TP and Chla among all groups. However, despite the TP concentration being significantly low in G3a, the subgroup has a Chla concentration almost as high as G3b. Also, it has a higher pH level than G3b.

The last group, G4, primarily found in sites with stream orders 7 and 8, appears to be related to WT, pH, DO and Chla in low discharge as primary factors rather than pollution-related parameters, unlike other groups. This group has two subgroups G4a and G4b, widely differentiated by contrasting responses in the four variables. For example, G4a is featured with high WT, and a high level of pH, DO and Chla in low discharge despite the low level of nutrients, while G4b has low WT, a low level of pH and DO, and the lowest Chla concentration in the lowest discharge.

6.3.2.2 Identification of indicator species for diatom groups

One or more indicator species per group were identified to characterise the attribute of the classified diatom groups. The indicator species analysis on the GOVD dataset returned the list of diatom taxa per group with their statistical probabilities (Table 6.1). According to the criteria ($stat \geq 0.5$ and $p < 0.05$) set in the Methods section, 26 taxa (in bold in Table 6.1) were selected as indicator species representing the diatom groups in the GOVD dataset.

Group G1 has a single species that fulfils the conditions. *Achnanthes minutissima* occurs in 64.3 % of sites belonging to this group, but nearly all sites ($B = 0.9743$) belonging to this group include the species. On the other hand, more than 70 % of sites with the presence of *Achnanthes alteragracillima* and *A. minutissima* var. *jackii* belong to this group, showing a good probability to be in G1, but the two taxa are not enough to be considered as indicator species owing to too low presence within the group. Similarly, *Fragilaria pinnata* and *Stephanodiscus parvus* have a too low presence in sites belonging to this group.

The second group G2 has two species *Nitzschia inconspicua* and *Navicula subminuscula* within the criteria as indicator species. *N. inconspicua* can be used to represent G2 as it is largely restricted to this group ($A = 0.7317$) and all sites belonging to this group include the species. *N. subminuscula* has nearly half the chance of belonging to this group, but 87.2 % of sites belonging to this group contain the species. However, *N. perminuta* is slightly below the criteria to become indicator species.

In G3, subgroup G3a has five species — *Navicula minima*, *Achnanthes subhudsonis*, *Nitzschia amphibia*, *Amphora pediculus* and *Cyclotella pseudostelligera* — that can be used to represent subgroup G3a. *N. minima* occurs in almost all sites belonging to this group, although the species has a 66.4 % possibility of belonging to this group. *A. subhudsonis* is largely restricted to this group ($A = 0.8293$) and nearly 70 % of sites belonging to this group contain the species. In contrast, *N. amphibia*, *A. pediculus* and *C. pseudostelligera* has a fairly low predictive value for the group, but sites with their presence in the group are at least over 65 %. In the meantime, although *Achnanthes exigua* and *Navicula seminulum* do not meet the criteria, their values for either component are greater than a half, suggesting that they might have the potential to represent this subgroup. In subgroup G3b, four species — *Nitzschia palea*, *Nitzschia fonticola*, *Navicula gregaria*, and *Fragilaria elliptica*

— are selected. The first three species are observed in more than 62 % of sites belonging to this group, however, their predictive values are quite low, remaining below half. *F. elliptica* has slightly over a half chance for both components.

Moving onto G4, subgroup G4a has 12 taxa that can be used to differentiate this group from the others — *Fragilaria construens*, *F. construens f.venter*, *F. construens f.binodis*, *F. crotonensis*, *F. capucina*, *F. capucina* var. *mesolepta*, *F. capucina* var. *vaucheriae*, *Aulacoseira granulata*, *Cocconeis placentula*, *Melosira varians*, *Navicula viridula* var. *rostellata*, and *Cymbella affinis*. The feature of indicator species for this group is the dominance of *Fragilaria* spp. that present at least more than 63.5 % of predictive value for the group. Of which, *F. construens* is the most representative with over 91 % chance of belonging to this subgroup. On the other end, *F. capucina* var. *gracilis* is not selected as indicator species since sites belonging to G4a are below a third, although it has a high value for component A. Besides, *A. granulata*, *C. placentula*, *M. varians*, *N. viridula* var. *rostellata* and *C. affinis* are selected as indicator species owing to 57–77 % of sites belonging to this group, but their predictive values are below half. Subgroup G4b has two species — *Cocconeis placentula* var. *lineata* and *Achnanthes convergens* — to define the feature of this subgroup. Sites that contain *C. placentula* var. *lineata* are largely restricted to this subgroup ($A = 0.7786$) and sites belonging to this group with the presence of the species are high with a 75 % chance. *A. convergens* has a 66 % chance of indicating this subgroup and its presence within the subgroup is 83 %. Meanwhile, *C. placentula* var. *euglypta* and *Nitzschia frustulum* have an over 60 % chance of indicating this subgroup, but their presence within the subgroup is too low to be considered as indicator species.

Table 6.1: Results of Indicator value analysis for the GOVD dataset. Taxa in bold are 26 of indicator species that meet the criteria ($stat \geq 0.5$ and $p < 0.05$).

Group (k=6)	*Species*	*A*	*B*	*stat*	*p.value*
G1	***Achnanthes minutissima***	0.6429	0.9743	0.791	0.001***
	Achnanthes alteragracillima	0.7994	0.2820	0.475	0.001***
	Fragilaria pinnata	0.5605	0.3269	0.428	0.003 **
	Stephanodiscus parvus	0.4163	0.1987	0.288	0.027 *
	Cocconeis pediculus	0.1882	0.3782	0.267	0.619
	Gomphonema angustum	0.4995	0.1410	0.265	0.089
	Achnanthes minutissima var. jackii	0.7398	0.0769	0.239	0.029 *
G2	***Nitzschia inconspicua***	0.7317	1.0000	0.855	0.001***
	Navicula subminuscula	0.4789	0.8723	0.646	0.001***
	Navicula perminuta	0.4569	0.5000	0.478	0.001***
	Achnanthes lanceolata	0.2520	0.7660	0.439	0.034 *
	Reimeria sinuata	0.3313	0.5745	0.436	0.003 **
	Navicula atomus	0.4025	0.2447	0.314	0.024 *
G3a	***Navicula minima***	0.6643	0.9615	0.799	0.001***
	Achnanthes subhudsonis	0.8293	0.6923	0.758	0.001***
	Nitzschia amphibia	0.3820	0.9038	0.588	0.001***
	Amphora pediculus	0.4141	0.6538	0.520	0.001***
	Cyclotella pseudostelligera	0.3531	0.7115	0.501	0.001***
	Achnanthes exigua	0.3370	0.6538	0.469	0.001***
	Navicula seminulum	0.6604	0.3269	0.465	0.001***
	Nitzschia paleacea	0.4288	0.3654	0.396	0.002 **
	Navicula cincta	0.3112	0.4423	0.371	0.007 **
	Gomphonema clevei	0.2081	0.5192	0.329	0.231
	Amphora veneta	0.1908	0.3654	0.264	0.278
G3b	***Nitzschia palea***	0.4413	0.8356	0.607	0.001***
	Nitzschia fonticola	0.3976	0.7558	0.548	0.001***
	Navicula gregaria	0.4796	0.6197	0.545	0.001***
	Fragilaria elliptica	0.5413	0.5164	0.529	0.001***
	Fragilaria ulna	0.4209	0.5915	0.499	0.027 *
	Stephanodiscus hantzschii	0.6345	0.3051	0.440	0.003 **
	Cyclostephanos invisitatus	0.5660	0.3286	0.431	0.003 **
	Cyclotella meneghiniana	0.3122	0.5915	0.430	0.011 *
	Nitzschia dissipata	0.6319	0.2394	0.389	0.002 **
	Cyclotella stelligera	0.3845	0.3239	0.353	0.058
	Navicula cryptocephala	0.2625	0.3708	0.312	0.429
	Diatoma vulgaris	0.2559	0.2770	0.266	0.800
	Navicula contenta	0.9407	0.0751	0.266	0.095
	Navicula amphiceropsis	0.3858	0.1361	0.229	0.262
	Navicula goeppertiana	0.5278	0.0892	0.217	0.289
	Gomphonema minutum	0.6160	0.0751	0.215	0.363
	Navicula mutica	0.8813	0.0516	0.213	0.399
	Fragilaria fragilarioides	0.7123	0.0563	0.200	0.568
	Navicula recens	0.4939	0.0563	0.167	0.322
	Navicula minuscula var. muralis	0.9807	0.0234	0.152	0.205
	Fragilaria robusta	0.7912	0.0140	0.106	0.931
G4a	***Fragilaria construens***	0.9102	0.9778	0.943	0.001***
	Fragilaria capucina	0.7538	0.9111	0.829	0.001***
	Fragilaria crotonensis	0.8436	0.8000	0.821	0.001***
	Fragilaria construens f. venter	0.6356	0.9778	0.788	0.001***
	Fragilaria construens f. binodis	0.6782	0.8000	0.737	0.001***
	Fragilaria capucina var. mesolepta	0.7034	0.6444	0.673	0.001***
	Fragilaria capucina var. vaucheriae	0.6842	0.6000	0.641	0.001***
	Aulacoseira granulata	0.4832	0.6667	0.568	0.001***
	Cocconeis placentula	0.5022	0.5778	0.539	0.001***
	Melosira varians	0.4269	0.6667	0.534	0.001***
	Navicula viridula var. rostellata	0.3475	0.7778	0.520	0.002 **
	Cymbella affinis	0.3746	0.6667	0.500	0.001***
	Fragilaria capucina var. gracilis	0.8012	0.3111	0.499	0.001***
	Gomphonema parvulum	0.3202	0.6889	0.470	0.006 **
	Aulacoseira ambigua	0.4450	0.4667	0.456	0.001***
	Navicula pupula	0.3866	0.5333	0.454	0.003 **
	Cymbella tumida	0.4852	0.3778	0.428	0.002 **
	Navicula cryptotenella	0.2461	0.6889	0.412	0.065
	Cymbella minuta	0.2225	0.6000	0.365	0.058
	Cymbella silesiaca	0.1997	0.6667	0.365	0.418
	Navicula decussis	0.3103	0.4000	0.352	0.024 *
	Aulacoseira alpigena	0.3924	0.2444	0.310	0.004 **
	Rhoicosphenia abbreviata	0.4944	0.1778	0.296	0.055
	Bacillaria paradoxa	0.4465	0.1778	0.282	0.103
G4b	***Cocconeis placentula var. lineata***	0.7786	0.7547	0.767	0.001***
	Achnanthes convergens	0.6655	0.8302	0.743	0.001***
	Cocconeis placentula var. euglypta	0.6106	0.3585	0.468	0.002 **
	Cymbella turgidula	0.3834	0.3207	0.351	0.139
	Nitzschia frustulum	0.7598	0.1321	0.317	0.026 *
	Cyclotella atomus	0.3629	0.2076	0.274	0.261
	Achnanthes japonica	0.4829	0.1321	0.253	0.021 *
	Achnanthes atomus	0.2942	0.0566	0.129	0.564

6.3.3 Relationships between environmental variables and diatoms

To find out the relationships between environmental variables and diatoms from 2009 to 2018 in the study area, Canonical Correspondence Analysis (CCA) was carried out. The results of the analysis are presented in Figure 6.7. It displays the ordinations of 13 environmental variables (12 types of water quality and discharge) and locations of 26 indicator species in CCA space. The first two CCA axes account for 40.9 % of all variances. The CCA1 axis (23.29 %) is strongly negatively correlated with TP and TN and weakly correlated with EC, discharge, and WT. On the other hand, the CCA2 axis (17.61 %) is strongly negatively correlated with Chla, COD, BOD and TOC. Besides, pH and DO are negatively correlated with the second axis in the opposite direction of SS and NH_3N.

The ordinations in the space explain relationships with indicator species as well as with the observations by diatom groups in the space (Figure 6.7 and 6.8). For instance, observations from G1 are mainly spread out along axis 1 on the right side of the space, suggesting that this group is featured with a low level of nutrients (e.g. TP and TN), low WT, low discharge, and EC. The sole indicator species *A. minutissima (ACH min)* for this group is located in the same direction. In contrast, observations from G2 are positioned on the opposite side of the G1 in the space, showing that they are positively related to nutrient pollution and SS. Two indicator taxa *N. subminuscula (NAV sub)* and *N. inconspicua (NIT inc)* positively correlate with SS. In the meantime, the most dominant diatom group G3 has distinctive distributions with subgroup G3a enclosed by G3b. Subgroup G3a is spatially confined in the narrow width of an ellipse situated in the direction of a high concentration of Chla and organic matter. Three indicator species *N. minima (NAV min)*, *C. pseudostelligera (CYC pse)*, and *N. amphibia (NIT amp)* are positively correlated with those variables, while *A. pediculus (AMP ped)* and *A. subhudsonis (ACH sub)* are more closely located to the direction of DO and pH in ordination space. The other subgroup G3b has observations scattered mainly at the centre of

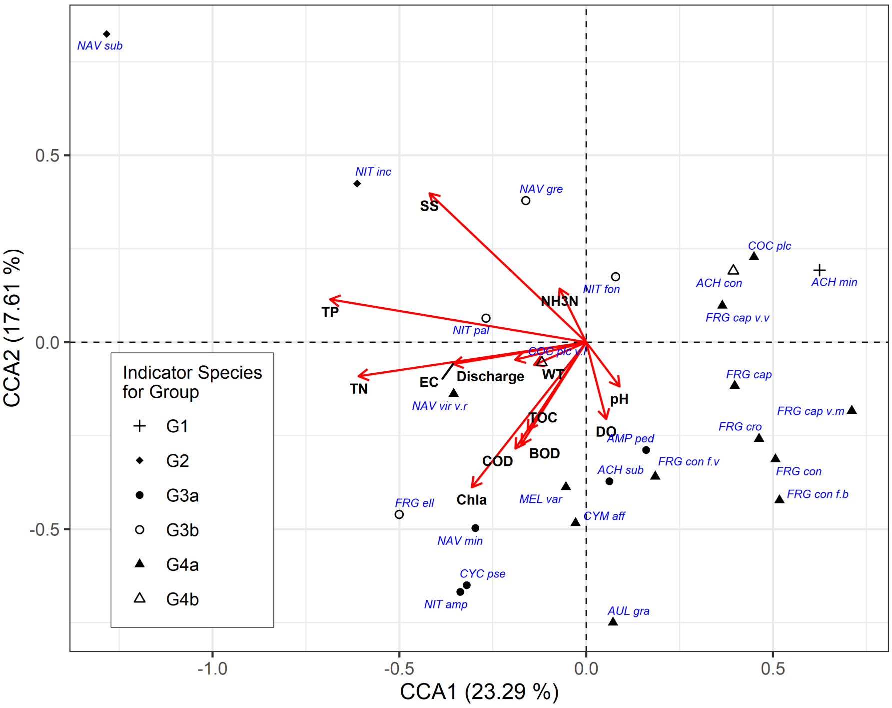


Figure 6.7: CCA biplots of ordinations and indicator species. Diatom taxa (in blue) on display are 26 indicator species.

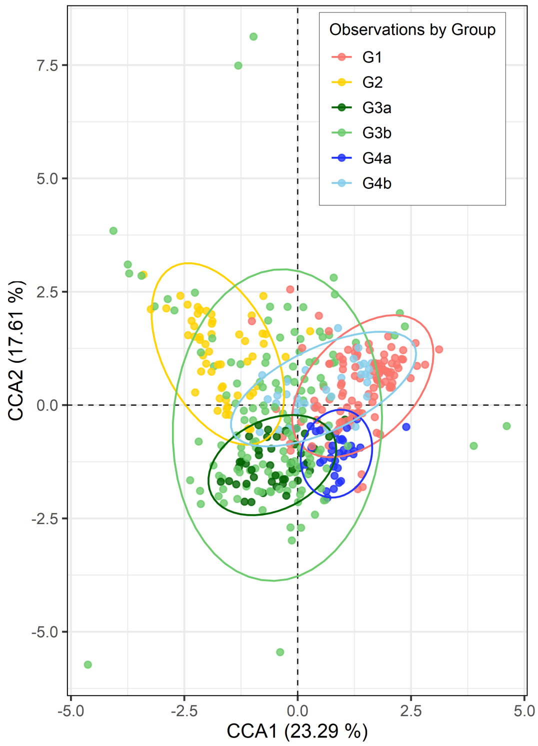


Figure 6.8: CCA plot of observations (site scores) by diatom groups

the space. Hence, the ellipse of this subgroup completely encompasses the ellipses of G3a and G4a, partially overlapping with those of G1, G2, and G4b. This G3b subgroup is nearly equally related to nutrients and organic matter. Accordingly, two indicator species *N. palea (NIT pal)* and *F. elliptica (FRG elt)* are correlated with nutrients and organic matter, respectively. However, *N. gregaria (NAV gre)* and *N. fonticola (NIT fon)* shows a weak correlation with those variables. Lastly, in G4, subgroup G4a, located in between G1 and G3a, is strongly correlated with a high level of pH and DO. Indicator species for this subgroup are *Fragilaria* spp., *M. varians (MEL var)*, *C. affinis (CYM aff)*, and *A. granulata (AUL gra)* broadly positioned in relation to the two variables. The other subgroup G4b largely overlaps with G1 and G3b, correlating with a low water temperature, discharge, and a low level of pollution. Indicator species for this subgroup are *A. convergens (ACH con)* and *C. placentula* var. *lineata (COC plc v.l)*. The former is closely positioned to indicator species *A. minutissima (ACH min)* from G1, while the latter is located at the centre of the space.

The observations in the CCA space were visualised by stream order as well as construction stages (Figure 6.9) to see what variables are most dominant and how variables have changed in connection with the construction of the barrages. Construction stages are divided into three stages — the Pre-Construction period for observations in 2009, the During-Construction period for observations between 2010–2011 and the Post-Construction period for observations between 2012–2018.

Figure 6.9 shows contrasting patterns of ellipses by the construction stage across all stream order sites, although an ellipse for the Pre-construction period at stream order 6–8 is not able to be generated due to insufficient observations. In particular, sites with stream order 9 show considerable changes in an ellipse by completely changing the major axis of an ellipse from the Pre-Construction, to the During-Construction, and Post-Construction periods. These observations are all recorded in sites from the reaches of the main river controlled by the barrages. These changes in an ellipse in these sites mean that these sites are significantly affected by the

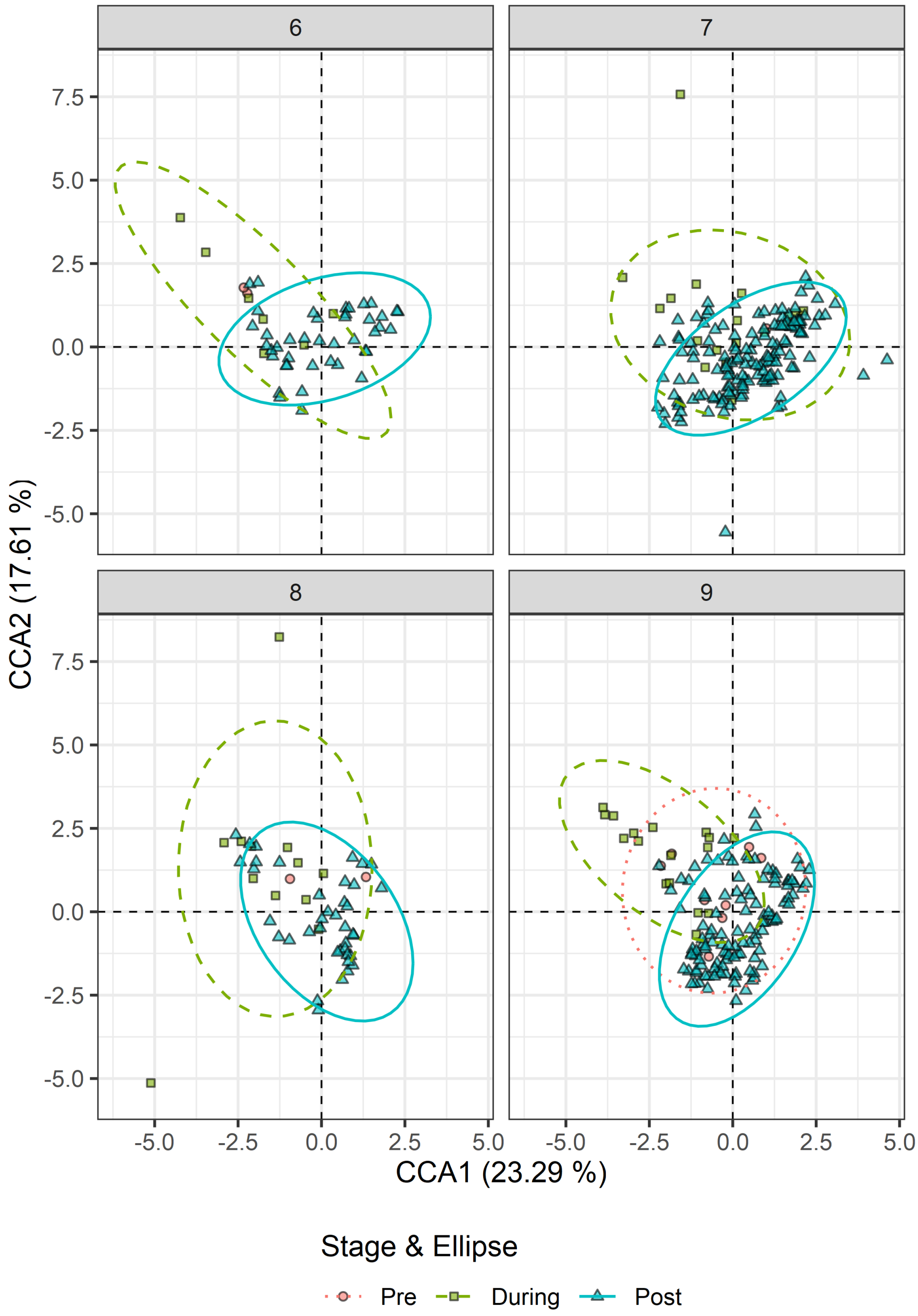


Figure 6.9: CCA plot of observations (site scores) by stream order and construction stage. Construction stages related to the barrages were divided into three stages; Pre-Construction (2009), During-Construction (2010–2011) and Post-Construction (2012–2018). Ellipses are 95 % confidence level and ellipses for Pre-Construction in 6–8 stream order were not generated due to insufficient observations.

construction of the barrages with the change of major environmental variables from a nutrient and SS dominating state in the Pre-Construction period to a SS dominating state in the During-Construction, to an organic matter, Chla, WT, and discharge dominating state in the Post-Construction period. At the same time, an ellipse in sites with stream order 9 decreased in the area after the construction. This result means that observations in the Post-Construction period tend to be more similar in the range of the environmental variables.

Sites with stream order 6 show a similar change from the During-Construction to the Post-Construction. This means that their prevalent environmental variables also changed due to the disruption. However, it is not possible to conclude that the impact of the construction was permanent to change the environmental variables in the Post-Construction period because of the lack of observations in the Pre-Construction period. In sites with stream orders 7–8, ellipses have changed in shape and angle from the During-Construction period to the Post-Construction period, but the extent of changes is less significant. This result means that the construction did not make an impact as much in those sites as in sites with stream order 9.

To summarise the findings, G1 and G2 have completely different attributes; G1 is related to less pollution and low water temperature, while G2 is closely correlated with SS and a high level of nutrient pollution. Subgroup G3a has a well-clustered distribution in the narrow space of the CCA space, aligned in the direction of organic matter and Chla. Subgroup G3b shows a wider distribution in the space, completely encompassing G3a and G4a and partially overlapping with G1, G2, and G4b. In contrast, G4a is positively related to pH and DO, while G4b mostly overlaps the distribution of G1. With regard to the construction of the barrages, sites with stream order 9 have significantly changed their relationships with environmental variables, shifting from a nutrient and SS dominating state to an organic matter and Chla dominating state. Sites with stream order 6–8 also have similar changes, but their changes are not as significant as in sites with stream order 9.

6.4 Discussion

6.4.1 What does the spatio-temporal distribution of diatom groups mean?

There are various frameworks proposed to conceptualise the structure of riverine ecology in space, spanning from having continuous and downward gradients (e.g. the RCC, Vannote et al., 1980), to having discontinued and patchy mosaics (e.g. Pringle et al., 1988), to being controlled by hydro-geomorphic processes at catchment or reach scales (e.g. the RES, Thorp et al., 2006). In Chapter 3, this study suggests the possibility of the downward structure of riverine ecology or the structure of six groups divided at the reach scale based on the distribution of centroids in the main river for water quality data (see Figure 3.25).

According to the classification of diatom groups combined with the water quality data, the Nakdong river had downstream gradients by the level of pollution from G1 in less polluted water, to G2 in polluted water, to G3 in severely polluted water between 2009–2010 (Figure 6.3). However, the structure of the six groups at the reach scale is not identified at this spatial scale. The results in the current study can be compared with the nationwide study (Hwang et al., 2011) that examined the distribution of benthic diatoms in South Korea in the autumn of 2009. At the national scale, Hwang et al. (2011) classified diatoms in the Nakdong river catchment into three major groups with strong downstream gradients from G1a and G1d in mountainous streams to G4 in lowland rivers. Despite the different spatial scales, the two subgroups in mountainous streams from the study of Hwang et al. (2011) are similar to a combination of G1 and G4b from the current study, while G4 in lowland rivers is similar to a combination of G2 and G3 identified in the current study in terms of indicator species. In addition, this project further identifies the occurrence of G4b at the regional scale in M1 and M5 in 2009. This subgroup is very similar to G1 from this project in terms of having a low discharge and low water temperature. For instance, the results of CCA (Figure 6.7 and 6.8) show that the ellipse of G4b overlaps with G1, while indicator species *Achnanthes convergens (ACH con)* from G4b and *A. minutissima (ACH min)* from G1 are closely located in a similar position of CCA space (Figure 6.7). However, the national-scale study of Hwang et al. (2011) did not discover G4b in the 2009 data possibly due to the different spatial scale.

Furthermore, the current project reveals the temporal changes of diatom groups from 2009 to 2018 in the study area. These developments are vital to comprehending how diatoms have responded to disturbances such as the construction of the barrages and the hydrological drought. One notable feature in the development of diatom groups in the main river is the disappearance of G2 in the upstream from autumn 2010, interrupted by the expansion of G3b into the upstream and more frequent occurrence of G3a. These shifts appear to be related to the construction of the barrages in the main river because the timing of these changes in the development of diatom groups is in harmony with the actual starting point of the construction that this project has inferred in Chapter 3. Despite the administrative reports (Kim, 2009; The FMRRP Committee, 2014) stating the onset of the construction in the summer of 2009, the construction had not started yet at least before early 2010 based on the visual examination of SS trends (Figure 3.21) and the water level records at the barrages (Figure 3.8). Google Earth satellite images also show that the Nakdong river was in a pre-construction state at the end of 2009 with sediments in the river channel (Google Earth Pro, 2009).

Based on these inferences and evidence, we confidently conclude that diatom groups in the entire year of 2009 would have been sampled before the construction and the downward gradients of water quality in the main river with a transition from G1 to G2, to G3b in the diatom groups are representative of the pre-construction state. In addition, the divide between G2 and G3b at M8–M9 in spring 2009 is thought to be associated with the most polluted water quality at M10, which is demonstrated in the distribution of centroids in the PCA results (see Figure 3.25).

Afterwards, by around spring 2010, the expansion of G3b initiated and replaced G2 in the upstream of the main river. These changes appear to be driven by the onset of the construction of the barrages in the main river in 2010 at the earliest. When the construction was completed in April 2012, the impact of the barrages continued as the main river hardly recovered to G2 in diatom groups until 2018, but had seen a more frequent occurrence of G3a on the top of G3b and G1 compared to the pre-construction period. Subgroup G3b is characterised with a high level of organic pollution, nutrients and Chla in large discharge (Figure 6.5) and is dominant in the downstream before the construction (Figure 6.3). Thus, the expansion of this subgroup into the upstream indicates that water quality and discharge in the upstream of the main river became similar to those in the downstream. The frequent occurrence of subgroup G3a is closely related to the effect of the barrages on the flow regime. This subgroup has a TP concentration as low as G1, nevertheless, it has a high Chla concentration with alkaline water, and high DO. These water quality features agree with the increasing trends of pH, DO, and Chla in the upstream of the main river due to the enhanced photosynthesis following slow-flowing water (see the trends in sites M4–M8 in Figure 3.18 and 3.22). As a result, this subgroup occurs more often in the upstream of the main river after the construction, but not in the tributaries (Figure 6.3). In 2018, the main river no longer had the downward gradient observed in 2009 but was dominated by G3a and G3b at the majority of sites.

In contrast, the change of diatom groups in the tributaries appears to remain independent in most sites and unchanged in time, whilst some sites experience a similar change as in the main river (Figure 6.3). For example, sites (e.g. T3b, T4, T5a, T5b) went through the conversion to G3b in around 2011, however, other sites (e.g. T6a–T7b) swiftly recovered to G2 with the intermittent occurrence of G3b. Also, the occurrence of G3a in the tributaries did not increase at all. Hence, the impact of the construction of the barrages on water quality and diatom assemblages in the tributaries was temporary and geographically restricted.

Meanwhile, this project also identifies the existence of subgroup G4a in temporal aspects of the distribution of diatom groups (Figure 6.3), which is closely related to the hydrological drought events in 2015 and 2017 in the study area (Figure 3.5 and 3.6). Similarly, the CCA results (Figure 6.7) support that observations from G4a are positively distributed along with ordinations of pH and DO, but have a negative correlation with discharge. This result reaffirms the excellent response of diatoms to hydrological conditions in rivers. For instance, in the literature, several studies report that hydrological drought events led to homogenisation in diatom assemblages in rivers (Peszek et al., 2021) and lakes (Klamt et al., 2020) regardless of their precedent diatom groups. In the Nakdong catchment, the emergence of G4a is more well-evidenced in the tributaries than in the main river. Also, the switch to G4a does not occur in sites between the barrages in the main river except M4 and M14. These are because of the role of the barrages that maintain a certain level of water in the main river despite the meteorological drought (see the water level in the dry year of 2015 in the barrages in Figure 3.8), whilst the tributaries were more exposed to the drought (Figure 3.6), resulting in more sites converting to G4a in diatom groups. Those sites converting to G4a in the tributaries in 2015 were mainly from the north region of the study area in which the meteorological drought event was much more severe than in the south. The response of diatoms to the hydrological drought is well-recorded as distinctive movements at stages 13–14 in the time-series trajectories in NMDS results (Figure 5.22).

At an intra-annual time scale, the distribution of diatom groups reveals the seasonal change of diatom groups as some sites (e.g. M8, M10, T6a, T11a) regularly switch from one group to another between spring and autumn (Figure 6.3). In particular, these seasonal changes are more obvious in the tributaries than in the main river with the recurring G1 group in spring and the prime example is T11a with a constant change between G1 and G3b. This pattern suggests that there is a possibility that certain species seasonally thrive and decrease at an intra-annual cycle, and the identification of those species can be useful in understanding the response of diatoms and ecological health assessment. For example, Snell et al. (2019) captured a robust and recurring seasonal change in stream diatom assemblages with a ratio of dominant species *Achnanthidium minutissimum* to *Amphora pediculus*, and *Gomphonema parvulum* to *Navicula lanceolata* in low order streams in the U.K. and demonstrated a novel way for the ecological quality assessment using the ratio. However, there is no similar study conducted in detail yet about the seasonal change of diatom taxa in Korea, although Choi et al. (2019) identified seasonal changes of the most dominant and second most dominant species in tributaries NS, YG, and WE in the study area by comparing samples taken in May, August, and October 2016. In their study, the taxa identified are somewhat different by rivers but are all common species and their relative abundances change throughout the year — *Achnanthes minutissima*, *Achnanthes convergens*, *Navicula subminuscula*, *Navicula minima*, and *Nitzschia inconspicua*. Their findings and evidence of the seasonal changes in this research suggest that there is a high chance that a seasonal baseline of diatoms would exist in the study area. However, it may be difficult to identify a seasonal change of diatoms in the main river as the seasonal variability between spring and autumn in the main river is weak throughout the whole monitoring period. Moreover, the thermal system in the main river has been modified by the barrages as demonstrated in the CCA results with the MYD dataset (Figure 5.16). Nevertheless, it is much needed to look into the response of diatoms to seasonality for better usage of diatoms in water quality assessment.

6.4.2 How useful are indicator species in determining diatom groups?

The selected 26 taxa as indicator species are all common diatom species found in streams or rivers in South Korea. Their ecological traits are primarily well-matched to the feature of water quality per group as well as previous findings in the literature. For instance, *Achnanthes minutissima* from G1 is one of the most commonly found species in Korean rivers with resistance to environmental turbulence (Hwang et al., 2011; Kim et al., 2013; Lee et al., 2016) and is often dominant in oligo–mesotrophic upland rivers and streams (Kelly and Whitton, 1995). This species is reported to have a positive correlation with forest land use and altitude, but a negative correlation with conductivity, BOD and nutrients (Hwang et al., 2011). These traits are similar to the feature of water quality in G1. In G2 where water is defined with a high concentration in nutrients, two indicator species *Nitzschia inconspicua* and *Navicula subminuscula* for this group are identified. In the literature, they are positively correlated with conductivity (Hwang et al., 2011), urban land cover (Walsh and Wepener, 2009), industrial complexes and municipal effluents (Pandey et al., 2018), and nutrients (Hwang et al., 2011; Tan et al., 2014; Choi et al., 2019).

Furthermore, the most dominant group in the main river, G3, has species that are resistant to a high level of pollution and their dominance is in agreement with the highest level of pollution among the groups in the catchment. For instance, in G3a, *Navicula minima* is reported to have a negative correlation with forest land cover (Hwang et al., 2011), but a positive correlation with urban area (Walsh and Wepener, 2009). *Nitzschia amphibia* and *Amphora pediculus* are frequently found in eutrophicated water with a high level of EC (Walsh and Wepener, 2009; Choi et al., 2019), while *Cyclotella pseudostelligera* is known to be resistant to pollution (Lee et al., 2016). Similarly, G3b has four species that are indicative of a high level of pollution. For example, *Nitzschia palea* is one of dominant species in polluted water and has a strong resistance to a high pollution (Lee et al., 2016; Klamt et al., 2020), to conductivity of 5,000–7,500 $\mu S\,cm^{-1}$ (Walsh and Wepener, 2009), and to a high TN concentration (Tan et al., 2014). *Nitzschia fonticola* is pollution tolerant species, easily found in rivers that are impacted by human influences (Lee et al., 2016; Kociolek, P., 2011). Besides, *Navicula gregaria* is an indicator species for eutrophication with a strong resistance to a high pollution (Walsh and Wepener, 2009; Lee et al., 2016), and *Fragilaria elliptica* thrives in water with a high conductivity (Choi et al., 2019).

The last group G4, closely related to WT, pH, DO and Chla in water quality, has two subgroups G4a and G4b, defined by contrasting ecological preferences. First of all, taxa in G4a are all related to alkalinity or mid-eutrophication except *Fragilaria construens* that is positively correlated with altitude (Hwang et al., 2011). Water quality in G4a is defined as a higher level of alkalinity, DO and Chla than in G4b (Figure 6.5) and the occurrence of G4a is temporarily restricted in 2005 in the north of the study area where the meteorological drought hit hard (Figure 6.3). Thus, the dry weather condition would enhance the positive feedback of photosynthesis activities in surface water in rivers, resulting in the increase of pH, DO, and Chla. In addition, the massive occurrence of *Fragilaria* spp. in this subgroup and their significance as indicator species appear to be evidence for the homogenisation in diatom assemblages during drought. For example, in the literature, Peszek et al. (2021) reported the rise of *Fragilaria gracilis* up to 47.2 % in abundance during drought in rivers in Magura National Park, Poland. This species which is regarded as a synonym of *F. capucina* var. *gracilis* (Guiry and Guiry, 2021) is also identified in the result of indicator value analysis (see G4a in Table 6.1) but is not selected by marginal deficit for the criteria. Nevertheless, the classification of this species into this subgroup supports its relation with meteorological dry conditions and its positive feedback on those variables. In contrast, G4b has two taxa that prefer a nutrient-poor condition and low conductivity. *Cocconeis placentula* var. *lineata* shows a negative correlation with nutrients (Choi et al., 2019), while *Achnanthes convergens* has a positive correlation with forest land cover and current, but a negative correlation with agricultural land cover and conductivity (Hwang et al., 2011).

Putting these interpretations together to understand the spatio-temporal pattern of diatom groups, the expansion of G3b in the main river with the more frequent occurrence of G3a indicates the dominance of diatom taxa that thrive in severely polluted water. These results are in agreement with the exacerbation of water quality in the main river after the construction of the barrages (Figure 3.20 and

3.22). The identification of G4a in 2015 is indicative of the effect of the hydrological drought, affected by the meteorological drought in 2015.

Meanwhile, the application of the criteria ($stat \geq 0.5$) to the result of indicator value analysis misses out on some important taxa that are identified as indicator species in other previous research. Although those missed taxa are primarily in agreement with the feature of water quality, this may have arisen due to different responses to seasonality or other environmental variables. Thus, this evidence suggests that the identification of indicator species is not always useful at inter-annual scales and it may become less powerful in the future due to the effect of climate change. For instance, *A. alteragracillima* from G1 is known to have robust negative correlations with conductivity, BOD, and nutrients (Hwang et al., 2011; Choi et al., 2019) and these traits are in line with the feature of water quality in G1. However, this species is only dominant in May samples, and second-dominant in August samples, but is not dominant in October samples in tributary YG (Choi et al., 2019). This example demonstrates that seasonality affects the occurrence of some diatom taxa. In group G2, *A. lanceolata* and *R. sinuata* show a low chance of belonging to the group ($A = 0.252$ and $A = 0.331$), despite their high presence within the group. Those species may be related to water temperature as Kim and Lee (2017) discovered consistent presences of both taxa at altitudes > 200 m regardless of the season in mountainous streams in the north of South Korea. Besides, *Navicula perminuta* from G2 is reported to thrive as the second dominant species following indicator species *N. inconspicua* in lower streams in Korea (Kim et al., 2013), meaning that they may share similar ecological traits. However, it is not selected as indicator species by marginal difference ($stat = 0.478$). Similarly, *A. exigua* and *N. seminulum* from G3a are indicative of a high level of nutrients in previous studies (Pandey et al., 2018; Choi et al., 2019), but are not selected for the same reason. These examples suggest that the application of indicator species analysis may miss some important taxa out at intra and inter-annual scales. They also emphasise the need to understand seasonal changes in diatom assemblages in rivers. Hence, temporal changes in diatom assemblages in rivers should be investigated and these will be vital to developing a better way for robust river health assessment.

6.4.3 What are the main environmental factors that control diatoms in the Nakdong catchment?

The results of CCA (Figure 6.7) demonstrate that nutrients (P and N), SS, Chla, and organic matter (BOD and COD) are important variables in explaining the distribution of observations from 2009 to 2018 at the Nakdong catchment scale. These findings are similar to those of previous research (Lobo et al., 1995; Tan et al., 2014; Hwang et al., 2011; Jung et al., 2016; Lee et al., 2021) in terms of the emphasis on nutrients and organic matter for determining diatom assemblages. For instance, in the nationwide study on the distribution of benthic diatoms in Korea, Hwang et al. (2011) revealed that chemical variables (EC, BOD, and nutrients) are important factors along with altitude, land cover, and current velocity in explaining the nationwide distribution of diatom groups. Similarly, Lee et al. (2021) demonstrated that chemical variables (TN, TP, PO_4P, and EC) account for 42.4 % of variances in explaining the distribution of diatom assemblages in the Nakdong river, overwhelming hydrological-meteorological variables (11.3 %) and physical variables (17.4 %).

On the other hand, this study further identifies SS and Chla as the important variables in distinguishing G2 and G3a from the ten years of observations and they are closely related to the construction of the barrages. The identification of the two variables is because of the different timescale of data that the current study deals with ten years of diatom records (the GOVD dataset), whilst the previous studies (Hwang et al., 2011; Lee et al., 2021) used diatom samples taken once a year or four times within a single year. The results of CCA by stream order (Figure 6.9) demonstrate that environmental variables are different in time and space and have changed significantly through different stages of the construction.

For instance, sites with stream order 9 where the construction took place during 2010–2011, went through dramatic changes from the extensive distribution of observations in the Pre-Construction period, to the SS dominating distribution in the During-Construction period, and to the organic matter, Chla, discharge, WT, pH, and DO dominating distribution in the Post-Construction Period. With these changes in environmental variables, observations in the Post-Construction period tend to be more clustered in ordination space. This means that the construction of the barrages has turned the main river to be a more homogeneous and less dynamic riverine environment. It also indicates that nutrients or SS are no longer as significant as before in the main river, but those variables (e.g. organic matter and Chla, discharge, WT, pH, and DO) became important.

The significant changes in environmental variables in the main river would have led to different responses of diatoms in the main river at a local scale depending on their relative locations to the barrages (the upstream and downstream of the barrages). However, it is not possible to demonstrate it with the results of this research at the regional scale. For instance, Kim et al. (2013) found that diatom assemblages in regulated rivers were significantly different from those in unregulated rivers in lower streams in the western region of South Korea. Similarly, Krajenbrink et al. (2019) revealed that river impoundment by dams led to more clustered diatom assemblages at upstream sites that were significantly different to those at downstream in the U.K. However, despite significant changes in riverine environments in the main river, there is no comparative research carried out yet in regards to the barrages. Recently, Lee et al. (2021) looked at seasonal variations of diatom assemblages in the main river using samples taken after the construction, but their findings are not compared with samples from the Pre-Construction period. Therefore, it needs more research to disentangle the impact of the new riverine environment on diatom assemblages at a local scale.

Meanwhile, sites with stream orders 6–8 show some degree of changes, going through different construction stages, however, the extent of the changes is overall less severe or limited. In these sites, an ellipse for the Pre-Construction period could not be determined because of the limited number of observations, therefore, their pre-construction state could not be confidently confirmed. Nevertheless, observations in the Pre-Construction Period mostly fall on the ellipse of the Post-Construction period. In the During-Construction period, observations are slightly stretched out in the direction of SS in all sites, however, these changes are not substantially significant as in sites with stream order 9. Those temporary changes may be explained by temporary construction work such as weir or bridge construction that took place at a similar time as the barrages across the sites (Figure 3.28).

6.5 Conclusions

This chapter has looked at the relationships between water quality and the GOVD dataset and how their relationships had changed in time and space. As the first step, the GOVD dataset were divided into diatom groups based on their similarity and their spatio-temporal distributions were constructed. Afterwards, the feature of the obtained diatom groups and their patterns were understood through the statistical summary of water quality and the identification of indicator species. Finally, the relationships between the environmental variables and the observations were established to find the key variables as well as to understand the distribution of the diatom groups. Key findings are:

1. Ten years of the GOVD dataset from 2009 to 2018 are divided into four major diatom groups with the latter two having two subdivisions — G1, G2, G3 (G3a and G3b), and G4 (G4a and G4b). The spatio-temporal distribution of diatom groups has changed differently between the main river and the tributaries. The main river had a clear downward gradient of diatom groups from G1 to G2, to G3b in 2009 before G3b started to expand into the upstream of the main river in 2010, replacing G2. From 2012, with almost no G2 in presence, the main river largely consisted of G1 and G3b in diatom groups with

frequent occurrence of G3a. In contrast, the distribution of diatom groups in the tributaries is independent of rivers and does not show a similar change as the main river. Although some sites (e.g. T3b, T4, T5b, and T6) had the intermittent occurrences of G3b over time from 2011, most sites in the tributaries were defined with the alternating occurrence of G1, G2 or G4b between spring and autumn, and remain similar during the whole period. Meanwhile, G4b is mostly identified throughout the time in the upstream of the main river or in some tributaries in the north of the study area, while G4a have a limited distribution in time and space in relation to the hydrological drought. In the main river, G4a is observed in sites not regulated by the barrages, while its presence in the tributaries is restricted to sites in the north in 2015 and tributary NG.

2. The statistical summary of water quality and the identification of indicator species characterise the feature of the diatom groups. The identified 26 diatom taxa are well-pitched with the feature of water quality by groups as well as the previous studies. Group G1 represents less polluted water quality among all groups with a low level of nutrient pollution, a low concentration in Chla with a low discharge and has *Achnanthes minutissima* as indicator species. Group G2 is characterised by a high concentration of nutrients and a low level of organic matter and Chla. Indicator species for this group are *Nitzchia inconspicua* and *Navicula subminuscula*. Group G3 (G3a and G3b) is defined with the highest level of organic and nutrient pollution as well as a high concentration of Chla. G3b shows a higher level of pollution than G3a, while G3a has a high concentration of pH, DO, and Chla despite TP concentration being significantly low. Thus, G3a is the diatom group related to the effect of the barrages on the flow regime through more active photosynthesis, while G3b indicates the most polluted water quality. Both subgroups have indicator species which are resistant to a high level of pollution — *Navicula mimina*, *Achnanthes subhudsonis*, *Nitzschia amphibia*,

Amphora pediculus, and *Cyclotella pseudostelligera* for G3a, and *Nitzschia palea*, *Nitzschia fonticola*, *Navicula gregaria*, and *Fragilaria elliptica* for G3b. Finally, group G4 (G4a and G4b) appears to be related to WT, pH, DO, Chla, and discharge rather than pollution-related parameters. G4a is featured with low discharge, high pH, and a high concentration of Chla, while G4b has the lowest discharge, low pH, DO, and lowest Chla. Indicator species for G4a are *Fragilaria construens*, *Fragilaria capucina*, *Fragilaria crotonensis*, *Fragilaria construens f.venter*, *Fragilaria construens f.binodis*, *Fragilaria capucina* var. *mesolepta*, *Fragilaria capucina* var. *vaucheriae*, *Aulacoseira granulata*, *Cocconeis placentula*, *Melosira varians*, *Navicula viridula* var. *rostellata*, and *Cymbella affinis*, while G4b has *Cocconeis placentula* var. *lineata*, and *Achnanthes convergens*.

3. Putting the results together, a downstream transition from G1 to G2, to G3b in the main river in 2009 is shaped by increasing pollution levels downstream. The expansion of subgroup G3b into the upstream of the main river from 2010 onward means that water quality and discharge in the upstream become similar to those in the downstream due to the barrages. Afterwards, the main river primarily remains dominant with G1 and G3b with frequent occurrence of G3a. This subgroup becomes common in the main river after the construction, which is the evidence of the effect of the barrages on water quality, discharge and diatoms related to enhanced photosynthesis following slow-flowing water in the main river. In contrast, some sites (e.g. T3b, T4, T5b, and T6) were temporarily turned into G3b from around 2011, but most sites in the tributaries remain similar over time with a seasonal change between spring and autumn. Meanwhile, the location of G4 is geographically restricted in the upstream of the main river or in the tributaries. Thus, it is related to hydrological drought or low discharge. In particular, the emergence of G4a coincides with the hydrological drought in 2015 temporarily and geographically that hit hard the north of the study area.

4. The results of CCA demonstrate that nutrients, SS, Chla, and organic matter are important variables in explaining variance in the Nakdong catchment from 2009 to 2018. With nutrients and organic matter already proved as the key variables in previous studies, the recognition of SS and Chla as key factors is new finding and the two variables are closely related to the construction of the barrages in the main river.

5. The results of CCA in connection with the construction stages of the barrages and stream order demonstrate different impacts of the construction on sites by stream order. Sites with stream order 9 have gone through the most abrupt changes from a wide distribution of ellipse in the Pre-Construction Period to an elongated shape in the direction of SS in the During-Construction Period, to a narrow width of an ellipse in the direction of organic matter and Chla in the Post-Construction period. These changes over time indicate that the construction of the barrages has switched the riverine environment in the main river to a more homogeneous environment, primarily controlled by organic matter and Chla. Accordingly, environmental variables organic matter, Chla, WT, pH, and DO became important in explaining water quality and diatoms in the main river. However, other sites tend to change little between the During-Construction and the Post-Construction period.

6.6 Summary

In this chapter, the spatio-temporal distribution of diatoms groups paired with water quality, discharge, and indicator spices and the relationships between environmental variables and diatoms were examined in relation to the construction of the barrages. These results provide a whole picture of river health in the Nakdong river and its tributaries from 2009 to 2018 and demonstrate the impacts of the barrages on riverine ecosystems in the rivers.

In the next chapter, I will compile all of the findings in Chapter 3–6 to address the research aim. Then, I will discuss inherent problems in the current use of diatoms in South Korea for water quality assessment based on the findings of this project. Finally, I will make suggestions for ways in which diatoms can be better used in water quality and river health assessments moving forward.

Chapter 7

Discussion

7.1 Introduction

This chapter brings all of the findings presented in Chapter 3–6 together, addressing the research aim by reviewing the changes in water quality and diatom assemblages during 2009–2018 in the Nakdong river and its tributaries. Also, it looks to find ways to utilise diatoms better in water-quality assessment in the future. Before doing so, key findings from each chapter are summarised in connection with the research questions (section 2.6.2 in Chapter 2). It then moves on to discuss what inherent problems are in the current use of diatoms and how diatoms can be better used in river health assessment in the future based on the findings. Finally, this chapter concludes with detailed suggestions drawn from the discussion.

7.2 Key research findings

Figure 7.1 presents the confirmed hydrological changes and the confirmed interactions in physical, chemical, and biological processes in the Nakdong river after the construction of the barrages based on the findings of this research. This project confirmed the creation of a reservoir-like water body in the main river with the increased water level, the increased water volume, the decreased flow velocity, and stratification since the completion of the barrages in April 2012. However, this research did not find any evidence of the increased groundwater level and its role in cooling water temperature which was expected in Figure 2.1. The changes in the hydrological systems cause lags in the cooling and heating processes and increase water temperature at the surface water in the river. This increase strengthens photosynthesis in the surface water, favoured by the dominance of dry conditions in the 2010s. The more active photosynthesis in slow-flowing water contributes to the increase in DO, pH, and Chla concentration. In the expected interactions, nutrients were expected to increase due to the disruption in longitudinal connectivity, however, the installation of sewage treatment plants and the introduction of the stricter criterion on phosphorus concentration contribute to the significant decrease in TP concentration, while TN level remains similar. After the construction, the SS level in the main river significantly dropped as expected. However, the low sedimentation after the construction did not help the water temperature drop. Key findings and detailed processes are summarised below by a chapter.

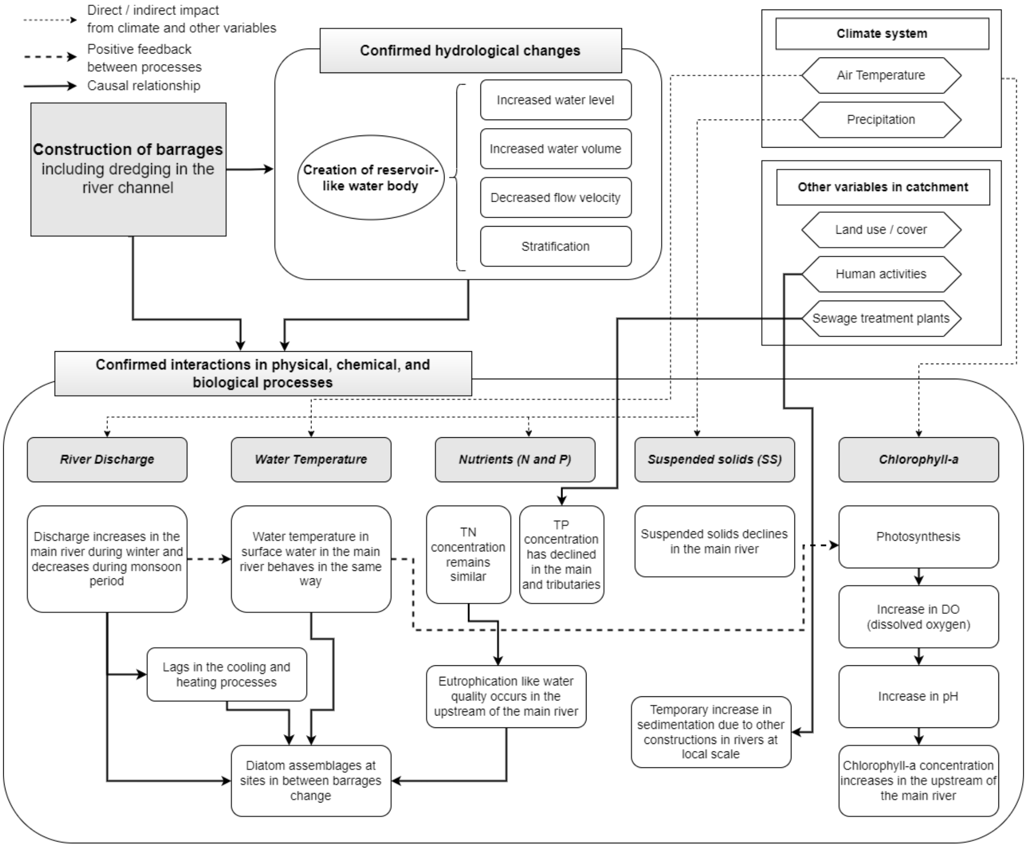


Figure 7.1: Confirmed hydrological changes and interactions in physical, chemical, and biological processes after the construction of the barrages based on the findings of this research

7.2.1 Changes in water quality in the Nakdong catchment

In Chapter 3, the changes in water quality from 2001 to 2018 in the Nakdong river and its tributaries were investigated in connection with changes in climate and hydrological systems. Accordingly, the hydrological system was examined by looking at precipitation at weather stations, discharge in the rivers, and hydrological data at the barrages. The trends of water temperature in the rivers were examined in relation to changes in air temperature and the construction of the barrages. Afterwards, the trends of water quality in the main river and the tributaries were understood with the identification of mean change points in the trends. Lastly, principal components in water quality data were found. The results of these analyses answer research question 1 and provide the basis for the diatom analyses.

In the Nakdong catchment, meteorological dry conditions are more prevalent in the 2010s than in the 2000s with two dry years identified in 2015 and 2017 (Figure 3.4) and those dry periods are well-presented in the response of the hydrological system. The impact of the droughts is more significant in the tributaries than in the main river as reflected in the reduction of discharge (Figure 3.5 and 3.6).

The change of flow regime was made in the main river when the barrages started operating in April 2012 by substantially reducing summer peak and increasing winter discharge that are demonstrated in the distribution of monthly discharge (Figure 3.7). Consequently, these hydrological changes in the main river would create a lentic environment with significant impoundment in the main channel. In contrast, there is no distinctive change to the flow regime detected in the tributaries unless it is meteorologically dry (Figure B.1–B.9).

Furthermore, the barrages have caused modifications to the thermal characteristics of water in the main river through impoundment. With the overall trend of water temperature in the rivers being primarily dependent on the change of air temperature in the catchment, sites M4–M15 in the main river started to behave in the same way around 2011 (Figure 3.13). The results of the Mann-Kendall test and the Pettitt test demonstrate that those sites have an upward trend with a change point between 2012–2013 (Figure 3.15 and Table 3.3). However, the tributaries do not show a similar trend. Provided that these changes are not identified in the tributaries where the barrages are not constructed, the identification of an upward trend after the change point between 2012–2013 is not a coincidence. It appears that the construction of the barrages has affected the water temperature in the main river. In contrast, the other sites in the main river (e.g. M1–M3) and all sites in the tributaries have remained independent in the behaviour of water temperature and are thus likely to be locally controlled (Figure 3.14).

The analysis of mean change points in the trends of water quality parameters identifies the presence of change points between 2012–2013 in the main river, which is most likely to be the barrages. The extent of changes is spatially different. The trends of pH, DO, COD, and Chla began to rise with a mean change point in 2012, and the increase is more apparent in sites M4–M8 in the upstream than those in the mid- and downstream of the main river (Figure 3.18, 3.19, 3.20, and 3.22). The increases in pH, DO, and Chla are attributed to slow-flowing water in the main river triggered by the barrages that contribute to the increase of retention time in flow (Noh et al., 2015), more active photosynthesis (Yu et al., 2014), and stratification of water (Wang et al., 2015; Jung and Kim, 2017). A lentic environment also contributes to the increase in COD concentration as increasing flow velocity promotes COD degradation in rivers (Tang et al., 2019). The trend of SS is affected by the construction of the barrages. During the construction of the barrages between 2010–2011, SS level significantly increased and remained high. In April 2012 when the barrages were completed, it started to decrease and dropped down below the pre-construction level in 2013 and remained very low in the post-construction period (Figure 3.21).

Meanwhile, TP concentration has been consistently on the decline in the main river and the tributaries since the early 2000s (Figure 3.23 and 3.17). Provided that the downward trend was already underway in both rivers before the beginning of the construction and this trend is also identified in the tributaries where the barrages were not constructed, it is unlikely that the barrages play a role in this decreasing trend. However, this significant improvement is attributed to the extensive efforts such as the enforcement of the stricter regulation on phosphorus concentration in released water and the installation of sewage plants across the basin (Figure 3.29 and 3.30). Traditionally, TP is regarded as the major determinant for Chla in the Nakdong river (Kim et al., 2021), however, the increase of Chla in the main river is inconsistent with the constant decrease of TP. This mismatch highlights the effect of the barrages on the increasing trend of Chla by causing a significant change to the hydrological system. Overall, the construction of the barrages in the main river has made significant impacts on hydrological and thermal systems as well as water quality in the main river, but not in the tributaries.

The PCA results show that BOD, COD and Chla are dominant variables as the principal components along with TN in the main river and TP in the tributaries (Figure 3.24 and 3.26). These findings are in line with the results of previous studies (Jung et al., 2016; Jung and Kim, 2017). Sites in the main river are aligned in geographical order along the first axis with the most polluted areas identified at sites M10–M12 because of the effect of tributary KH (Figure 3.25 and 3.27). The distribution of the centroids in the main river can be viewed as having continuous gradients of physical variables at the catchment scale (e.g. the River Continuum Concept, Vannote et al. 1980) or having a downstream array of patches at the reach scale by factors like hydrogeomorphic processes (e.g. the Riverine Ecosystem Synthesis (RES), Thorp et al. 2006). In the tributaries, a downward trend along the first axis is also identified between two sites within a river. These structures in the rivers can provide physical habitats and ultimately determine the distribution of diatoms. Hence, it is important to understand these structures.

7.2.2 Replicability of diatom assemblages

In Chapter 4, the replicability of diatom assemblages within a reach was investigated to answer research question 2. In water-quality assessment studies using epilithic diatoms, a practice of making a composite sample by merging diatoms collected from multiple rock samples is widespread, however, it has not been tested yet how similar diatom assemblages are to each other within a reach from random sampling. For this analysis, the ROD dataset was established on fieldwork in which five rock samples were each collected at sites A–C at the same time in the Nakdong river.

The result of NMDS analysis shows that the 15 samples are clearly dispersed between the sites in ordination space, but well-clustered within the sites with samples from site B more dispersed in space (Figure 4.4). Those three sites are characterised by different types of diatom taxa (Figure 4.5). Similarities of the 15 samples are demonstrated in the results of ANOSIM (Figure 4.7) that the null hypothesis that similarity between the sites is equal to those within the sites should be rejected. The analysis also demonstrates that there are more similarities within the sites than between the sites. Similarities within the sites range from 58.9 to 75.2 % with a mean of 67.8 % between the sites (Figure 4.8). Of the sites, site C has the highest similarities with 75.2 %, while site B has the lowest similarities with 58.9 %. Similarities at sites A and C and the mean similarity as a whole are sufficient for their samples to be considered replicate samples, while site B is below the threshold of 60 %, which is adopted from the study of Kelly et al. (2009).

Despite a targeted-habitat method (rock samples) being applied and the recommendations being followed as best as possible, variability in diatom assemblages at a reach scale still can arise because of the difference in physical, chemical, and biological characteristics at the local scale such as the influence of tributary or unintended errors in sampling process such as size or shape of rocks. Therefore, it is important to understand what variability means in water quality assessment.

The low similarity in site B mainly arises from a few diatom taxa in different abundance, identified in samples B2 and B4 compared to the rest of the samples in site B (Figure 4.6). For example, with *Achnanthes exigua* in high abundance in the rest of the samples in site B, B2 has a high abundance of *Fragilaria elliptica* and *Navicula seminulum*, whilst B4 has *Nitzschia amphibia* and *N. inconspicua* instead. However, their ecological traits in water are similar in relation to a high level of pollution, having similar sensitivity values for the mTDI. This means that the differences in diatom assemblages in site B would make little impact on the result of water quality assessment using the TDI due to a trade-off in the process. In addition, the occurrence of different diatom taxa in site B may have been caused due to different growth forms rather than the difference in water quality. Nevertheless, these differences in site B may cause a problem if the structure of diatom communities is directly compared using a dissimilarity index. Thus, it is recommended not to use a single sample within a reach for water quality assessment. Variability in diatom assemblages within a reach can be less influential when multiple samples within a reach are taken and made into a composite sample.

This research is the first attempt to examine the variability among random samples at a reach scale. It concludes that epilithic diatoms are reproducible at 67.8 % on average within a reach from multiple random rock samples. This result suggests that diatoms can be used as a good indicator for water quality assessment. However, it is little known about what level of similarity would be naturally expected to be observed on average from random sampling in time and space. Therefore, it is necessary to expand this experiment with a consideration of more sampling sites in rivers for better statistical significance.

7.2.3 Changes in diatom assemblages in the Nakdong catchment

In Chapter 5, the GOVD and MYD datasets were scrutinised 1) to evaluate the effectiveness of diatom-based indices, 2) to examine variability in diatom assemblages collected twice within the autumn survey period, and 3) to examine the change of diatom assemblages in the Nakdong catchment between 2009–2018 in relation to water quality, and the construction of the barrages. These analyses answer research questions 3–5.

First, the review of the mTDI results in the main river and the tributaries from 2009 to 2018 exhibits that river health in all catchments have been significantly improved from very bad or bad levels in the environmental grades to a satisfactory level or beyond (Figure 5.3 and 5.4). The improvement in river health is contradictory to the worsening trends of water quality such as BOD, COD, Chla, and EC in the main river since around 2012 (Figure 3.20 and 3.22). However, the index was developed based on the sensitivity of species to phosphorus concentration (Kelly and Whitton, 1995), and the significant decrease in TP concentration (Figure in 3.17) in the catchment since the early 2000s explains the improvement of river health. This means that the mTDI is an effective and powerful tool to monitor phosphorus concentration in water-quality assessment. The review of the mTDI results does not show any sign of fluctuation at the time of the construction of the barrages, or the hydrological drought, but remains independent during the period. This result means that the index is a robust means to assess water quality regarding changes in phosphorus without being influenced by such environmental stressors.

The application of three diversity indices — Species Richness, Species Evenness, and Shannon Diversity Index — to the GOVD dataset constructs the temporal changes of diversities in the catchment. The results display that diversity values are responsive to the construction of the barrages to a limited degree and the hydrological drought to some degree (Figure 5.6–5.8). Diversity in three indices all significantly increased in 2015 when the hydrological drought occurs and this result confirms the previous finding that three indices with diatoms are related to hydrological effects (Boix et al., 2010). In regard to water quality, diversity in three indices gradually increases in sites with stream order 9 where water quality is worsening since around 2012. This positive correlation is in line with the result of Tan et al. (2014), but a negative correlation between diversity and pollution is prevalent in the literature (e.g. Camargo and Jiménez 2007; Chen et al. 2016). These contradictory results in the application of diatoms with diversity indices support the study of Blanco et al. (2012) that revealed poor linear correlations between indices and trophic levels and discouraged the use of indices with diatoms. Unlike the expectation, diversity indices do not show any significant change in relation to the construction of the barrages. This may be attributed to sampling frequency and sampling effort. Diatoms have a short life span at a daily timescale, but the GOVD dataset consists of samples taken twice a year at an interval of five to six months, which would have allowed enough time for diatoms to recover and evolve unless physical disturbance from the construction was consistent or was close to the time of sampling. There is also a possibility that man-made errors may have been made for diatom identification due to analysts coming in and out every year. As a result, sampling timing should be specifically set for a targeted disturbance or sampling frequency should be increased to have more resolution of data when it comes to using diversity indices with diatoms.

Variability of diatom assemblages within the GOVD survey period was examined using the MYD dataset, consisting of two diatom samples per site collected three weeks apart in autumn. In the current use of diatoms in water quality assessment in South Korea, diatom sampling is carried out once each during the spring (May–June) and autumn (Sep–Oct) surveys in accordance with the recommendations (Kelly et al., 1998) as those periods have generally low and stable hydrological conditions in rivers. The comparisons of the MYD and ROD samples demonstrate that samples in the same sites from the two datasets are located in a similar position in the space. This result means that their relationships with each other remain preserved by responding to seasonal change. The results of comparisons (Figure 5.11) show that changes within the three week period in the MYD samples are bigger than the difference within the groups in the ROD samples but smaller than one between the groups in the ROD samples taken at the same time. This result means that diatom assemblages can change as quickly as within three weeks. Those changes in the MYD samples are not significantly different by river type (main river and tributary) or stream order, but different between sites affected by the barrages and not (Figure 5.12).

The results of CCA with the MYD dataset demonstrate that water temperature is the key factor explaining variance most within the three week period (Figure 5.15). This result highlights the important role of water temperature on diatoms in rivers. In water quality assessment, although Kelly et al. (1995) found no significant influence of season on the performance of indices, the role of water temperature and temporal change of diatoms are largely overlooked in the literature. Passy

(2007a) revealed that 51 % diatom variance is explained by the temporal and spatial datasets and their covariance with the environmental data, while the environmental data itself only explain 9 % diatom variance. Zou et al. (2018) also demonstrated that seasonal diatom succession was driven by water temperature, not trophic levels. These results show the need to investigate temporal changes of diatoms and it will be important in utilising diatoms in water quality assessment. The significant effect of water temperature within the GOVD survey period suggests that variability in diatom assemblages is susceptible under the current 'one visit per site within the two month period' sampling practice and this will likely increase more due to the effect of climate change. Therefore, it is recommended to make effort to minimise variability caused by seasonality or unusual climate condition by increasing sampling frequency by up to two visits per site at a minimum during the two months or reducing the two months to one month.

With water temperature being most important within the three week period, its impacts on diatom assemblages are not the same across the sites. The results of CCA by stream order (Figure 5.16) show that change is the least in sites with stream order 9 compared to other stream order sites. However, this result does not necessarily mean that stream order is the determinant factor. Sites with stream order 9 are all from the reaches of the main river in between the barrages, therefore, they respond least to the change of water temperature as a result of impoundment caused by the barrages. These findings indicate that the barrages in the main river have caused a modification to the thermal system through the change in the hydrological condition such as impoundment and lentic environment. These results are similar to those of previous studies (Krajenbrink et al., 2019; Peszek et al., 2021). Therefore, the change in the thermal system in the main river will affect the response of diatom assemblages.

In an attempt to examine the change of diatom assemblages in relation to the construction of the barrages and the hydrological drought in the Nakdong catchment, observations of diatom assemblages in the GOVD dataset were examined using

NMDS analysis and trajectory analysis. The results of NMDS analysis (Figure 5.21 and 5.22) show that sites are roughly categorised by their movements over time in NMDS space. Under stable and undisturbed conditions, sites (e.g. T8a, T8b, T11a, T11b) have remained in a similar position of NMDS space with seasonal switches between spring and autumn, but this is rare. In most cases, sites have moved and changed their positions in NMDS space and these sites are divided into two groups. Sites in the first group (e.g. M4–M10, M13–M15, T2, T4, T10) have completely relocated themselves to the other side of NMDS space, while sites from the second group (e.g. M1, M3, M12, T1, T3, T5, T6, T7, and T9) have returned and randomly moved around in a similar space after going through abrupt changes. Those sites from the first group are mostly from the main river where the construction of the barrages has altered the hydrological conditions, while sites in the second group are temporarily influenced by other construction works that were only able to cause a disturbance temporarily, but were not sufficient to alter the structure of diatom communities such as river embankment maintenance, or bridge or weir construction (Figure 3.28). These contrasting observations by these groups indicate that diatom assemblages in the main river have significantly changed due to the construction of the barrages. These results are in agreement with studies (Wu et al., 2009; Krajenbrink et al., 2019) looking at the response of diatom assemblages to dam construction.

Furthermore, the responses of diatom assemblages in the GOVD dataset were examined using Community Trajectory Analysis (De Cáceres et al., 2019) by measuring geometric vectors (length and angle) between observations in a time-series order and a survey-based order in NMDS space. The results of trajectory analysis are effective in providing a whole picture of responses of diatoms to seasonality, the construction of the barrages, and the hydrological drought during 2009–2018.

The distribution of the Time-series Segment Length (TSL) by stream order (Figure 5.24 and 5.25) show that TSL in sites with stream order 9 are significantly high during TSL_7–TSL_{11} before it gradually comes down and remains low. This phenomenon is only observed in those sites and appears to be related to the barrages. This interpretation means that the extent of changes in diatom assemblages was big two years after the completion of the barrages in April 2012 and their changes started to diminish from spring 2014 and remain small in the Post-Construction period. The impacts of the barrages are identified in the results of the Survey-based Segment Length (*SSL*) (Figure 5.26). They show that sites with stream order 9 had a big surge in *SSL* in both surveys 2012. This rise means that observations in 2012 significantly moved far away from their previous observations in NMDS space, which suggests that diatom assemblages changed significantly between 2011 and 2012. However, in other sites with stream orders 6–8, *TSL* remained similar or temporarily increased when the aforementioned constructions such as weirs or bridges were underway. In trajectory angle, the results of the Time-series Segment Angle (*TSA*) show that obtuse angles were dominant over time and there was no significant change in the angle at the time corresponding to TSL_7–TSL_{11}. This result means that the effect of seasonality remained similar during the high *TSL* period after the completion of the barrages. The results of the Survey-based Segment Angle (*SSA*) display that there was a sudden drop in the angle at SSA_{2014}Spr, while obtuse angles at around 100–120° were dominant over time. This drop coincides with TSL_{11} in spring 2014 and indicates that diatom observations in spring 2014 continued to move straight from two previous observations from 2012 and 2013. As of spring 2014, diatom observations in NMDS space completed their straight movement and started to settle on the other side of the space. These findings are similar to those of studies (Wu et al., 2009; Krajenbrink et al., 2019). In particular, Wu et al. (2009) presented the distribution of diatom assemblages in NMDS space using samples taken before, one year, two years and three years after dam construction. Their results display the movement and relocation of diatom observations with clustering patterns in ordination space two years after the construction.

The results of *TSL* and *SSL* also demonstrate the response of diatom assemblages to the hydrological drought in 2015. TSL significantly decrease at TSL_{14} in all sites (Figure 5.25), while SSL significantly increase at SSL_{2015}Spr and SSL_{2015}Aut. These responses mean that observations from spring and autumn in 2015 are similar to each other, while observations in the dry year of 2015 are significantly different to observations in the non-dry year of 2014 in both surveys. This result indicates that diatom assemblages during droughts become similar to each other regardless of season, but they are different to those in non-dry years. This is the effect of hydrological drought on diatoms as reported in other studies (Boix et al., 2010; Peszek et al., 2021).

The responses of diatom assemblages to the construction of the barrages and the hydrological drought in the Nakdong catchment highlight the importance of hydrology in determining the distribution of diatoms in rivers at the catchment scale.

7.2.4 Comprehensive understanding of changes in water quality and diatom assemblages in the Nakdong catchment

Chapter 6 attempted to comprehensively understand changes in water quality and diatom assemblages in the Nakdong catchment from 2009 to 2018 by linking water quality data to the GOVD dataset. The classification of the GOVD dataset into groups, the characterisation of the obtained groups by water quality, discharge, and indicator species and the establishment of relationships between environmental variables and diatoms were carried out. The results of these analyses answer research question 6.

The ten years of diatom assemblages in the Nakdong catchment were divided into four major diatom groups (G1–G4) with the latter two having two subdivisions a and b (Figure 6.2). The distribution of diatom groups (Figure 6.3) displays a different response in the main river from the tributaries in time and space. The main river had a clear downward gradient from G1 in the upstream, to G2, to G3b in the downstream in 2009. From autumn of 2010, the dominant G3b started to expand to the upstream of the main river and replaced G2. Afterwards, the main river had diatom groups consisting of G1 and G3b with a more frequent occurrence of G3a until 2018. In contrast, diatom groups in the tributaries are different by the site. Most rivers (e.g. T1–T6, and T8–T9) are defined with the alternating seasonal occurrence of G1, G2 or G4b between spring and autumn, while T7, T10, and T11 have mostly G3b in dominance. In addition, sites in the tributaries do not have a similar change of replacement in diatom groups but remain similar over time. In the meantime, subgroup G4b is identified in the upstream of the main river and the tributaries, while subgroup G4a is restricted in time and space; the occurrence of G4a was primarily identified in 2015 and the tributaries.

These spatio-temporal distributions and the changes in diatom groups are understood through the statistical summary of water quality and discharge data (Figure 6.5) and the identification of indicator species (Table 6.1). Overall, the feature of water quality per group is well-matched with ecological traits of indicator species per group and this demonstrates that diatoms are a good indicator for water quality assessment. For example, water quality in group G1 is characterised by a low level of nutrient pollution and a low Chla concentration compared to other groups. The indicator species for this group is *Achnanthes minutissima*, which is largely dominant in oligo–mesotrophic upland rivers and streams (Kelly et al., 1995). Group G2 shows a high concentration of nutrients and a high level of SS in water quality, which explains the discovery of *Nitzschia inconspicua* and *Navicula subminuscula* as indicator species. These two are reported to be positively correlated with conductivity (Hwang et al., 2011), nutrients (Hwang et al., 2011; Tan et al., 2014; Choi et al., 2019), urban land cover (Walsh and Wepener, 2009), and industrial complexes and municipal effluents (Pandey et al., 2018). In addition, the most dominant group in the catchment, G3 is defined with the highest level of organic and nutrient pollution as well as a high Chla concentration with the highest discharge. Within the group, subgroup G3b tends to have more polluted water than G3a. Instead, G3a is featured with more alkaline water, a high level of DO, and

Chla concentration despite TP concentration being significantly low. Accordingly, indicator species for these two subgroups are closely related to a high pollution and human influences or well-known for being pollution-tolerant (Walsh and Wepener, 2009; Lee et al., 2016; Choi et al., 2019); G3a has *Navicula minima*, *Nitzschia amphibia*, *Amphora pediculus Cyclotella pseudostelligera*, while G3b has *Nitzschia palea*, *Nitzschia fonticola*, *Navicula gregaria*, and *Fragilaria elliptica*. In contrast, Group G4 is more related to WT, pH, DO, and Chla in low discharge rather than pollution-related parameters. For instance, subgroup G4a, which mainly appears in sites where the hydrological drought hit in 2015, is featured with high WT, and high levels of pH, DO, and Chla. The indicator species for this subgroup are various (see Table 6.1), but the massive occurrence of *Fragilaria* spp. is most obvious. This result indicates the homogenisation in diatom assemblages during the drought as has been found in other studies (Klamt et al., 2020; Peszek et al., 2021). Subgroup G4b has low WT, a low level of pH, and DO, and low Chla in water quality and two species *Cocconeis placentula* var. *lineata* and *Achnanthes convergens* are identified as indicator species. They have a negative correlation with nutrients (Choi et al., 2019) and agricultural land use and conductivity (Hwang et al., 2011).

Putting these interpretations together, the expansion of G3b to replace G2 in the upstream of the main river indicates the exacerbation of water quality and the increase of discharge with the emergence of pollution-resistant species in the upstream of the main river following the construction of the barrages. These results are in agreement with the worsening trends of water quality with the presence of mean change points at the time of the construction of the barrages in the main river. In addition, the more frequent occurrence of G3a in the main river after the construction strongly supports the effect of the barrages on the flow regime through more active photosynthesis, which contributes to the increase in pH, DO, and Chla (Figure 3.20 and 3.22). The abrupt occurrence of G4a in the catchment results from the hydrological drought event that hit the study area in 2015. This result reaffirms previous findings (Klamt et al., 2020; Peszek et al., 2021) that extreme drought events homogenise diatom assemblages.

Finally, the results of CCA demonstrate that nutrients (P and N), SS, Chla, and organic matter (BOD and COD) are important variables in explaining the observations from 2009 to 2018 (Figure 6.7 and 6.8). The detection of nutrients and organic matter as the key variables is in line with previous findings in Korea (Hwang et al., 2011; Jung et al., 2016; Lee et al., 2021). Furthermore, this study identifies SS and Chla as key variables in distinguishing G2 and G3a from the observations and they are closely related to the construction of the barrages in the main river.

The breakdown of the CCA results by stream order (Figure 6.9) displays a different pattern of observations in relation to the construction stage of the barrages. For example, sites in stream order 9 show significant changes in the shape of an ellipse through the three stages. In the Pre-Construction period, they have an extensive distribution of observations at the centre of CCA space, slightly closely distributed in the direction of nutrient pollution. During the During-Construction period, they have more observations related to SS before they change close to the organic matter, Chla, discharge, WT, pH, and DO dominating distribution in the Post-Construction period. In addition, observations in those sites for the Post-Construction period are more clustered in a narrow area of an ellipse. These changes mean that the construction of the barrages has turned the main river to be a more homogeneous and less dynamic environment. Also, they indicate that nutrients or SS are no longer as significant as before, replaced by those variables in the Post-Construction period. In contrast, sites with stream orders 6–8 show some degree of changes in an ellipse in relation to the construction stage, however, the degree of changes is less significant. Although an ellipse for the Pre-Construction period could not be determined for these sites due to the insufficient number of observations, they have observations in the Pre-Construction period that fall on the ellipse of the Post-Construction period and thus they were not significantly different. Overall, the impact of the construction of the barrages in the tributaries is not as significant as in the main river.

7.3 How can diatoms be better used in river health assessment?

Based on the findings above, I discuss ways how epilithic diatoms can be better used in water quality and river health assessment moving forward.

7.3.1 The prospect of the use of the mTDI

The review of the mTDI results (Figure 5.3 and 5.4) in the study area shows that the environmental grades assessed by the index are in harmony with the time-series trends of TP concentration in water (Figure 3.17 and 3.18), without being influenced by any hydrological conditions in the rivers such as the construction of the barrages and the hydrological drought in 2015. In the Korean literature, several studies (Kim et al., 2017; Choi et al., 2019; Lee et al., 2021) generally report good performances of the index with nutrients (P and N) in the Nakdong river and some tributaries. This agreement is not a surprise as the index was developed based on the sensitivity of species to phosphorus concentration in water (Kelly and Whitton, 1995). Therefore, it is beneficial to continue to use the index in water-quality assessment when we want to monitor and assess water quality in relation to phosphorus concentration.

However, the use of the index in the Nakdong catchment is no longer as effective as it was in the past and the prospect for it is not likely to be promising moving forward. In the past, the mTDI was selected by the South Korean government on the assessment that nutrient pollution was more severe in Korean rivers than organic pollution (Choi, 2006; Noh et al., 2009). However, it is no longer as threatening as it was in the catchment with a gradual decrease in phosphorus concentration (Figure 3.17 and 3.23) owing to the efforts such as the implementation of a stricter criterion on phosphorus concentration limit and the installation of sewage treatment plants across the catchment (Figure 3.29). Although the use of the index would still be useful to monitor a TP level in rivers, nevertheless, other indices are needed that are more sensitive to the other pollutants such as organic matter in these environmental conditions in the Nakdong catchment.

Despite the capability of the mTDI to monitor the response of diatoms to phosphorus concentration in water, the application of the index for generic water quality assessment or holistic river health assessment is strongly discouraged because it may lead to confusion in decision-making for sustainable river health management. For example, the consistent increases in the mTDI scores and its environmental grades in the catchment (Figure 5.3) are contradictory to the worsening trends of COD and Chla in the main river (Figure 3.20 and 3.22). This disagreement demonstrates that the mTDI delivers only one aspect of water quality in river health assessment. Therefore, exaggerated interpretations of the mTDI results or too much emphasis on phosphorus concentration in river health assessment may give a false belief to local governments or decision-makers in policy-making for river health management. Hence, it is not recommended to use the index alone to assess water quality or river health. The holistic understanding and assessment of water quality and river health will be reached only if the mTDI is supplemented with other types of indices or a holistic approach to diatoms like this project.

7.3.2 What variables and/or species can be used better?

The construction of the barrages in the main river has caused significant changes in the riverine environments in the main river such as impoundment, increased water level (Figure 3.8), and regulated flow (Figure 3.9). As a result of these changes, organic matter (BOD and COD) and Chla became more influential as key environmental variables to explain the observations in the main river (Figure 6.7 and 6.9). This result suggests that these variables are now fast-rising threats in the main river to be targeted to monitor (Figure 3.20 and 3.22). Under these riverine conditions, the use of the mTDI is off-target and less effective as a tool to conduct water quality assessments in the main river.

In addition, the lentic environment formed by the barrages has significantly affected the thermal system in the main river as has been demonstrated in the analysis of the trend of water temperature and the variability with the MYD dataset. For example, the time-series trends of water temperature (Figure 3.13 and 3.14) show that water temperature (e.g. M4–M15) in the main river started to behave similarly with more close correlation at around 2011, while the tributaries remained different throughout the time. This different response of water temperature between the main river and the tributaries suggests the effect of the barrages on water temperature in the main river. Moreover, impoundment caused by the barrages has triggered a lag in the response of water temperature to air temperature in the main river, which then resulted in less change in diatom assemblages compared to others in the tributaries (Figure 5.15 and 5.16). This lag in the response of water temperature is similar to the impact of dams on water temperature (Long et al., 2016). These results all suggest that water temperature becomes more influential as a variable after the construction to affect the response of diatom assemblages. Besides, impoundment by the barrages has made the main river more favourable for the growth of Chla (Figure 3.22) by increasing the retention time of flow with the stratification of water in the reservoir of the barrages. In this project, the increase of water temperature, pH, DO, and Chla (Figure 3.18, 3.19, and 3.22) is identified in the sites (e.g. sites M4–M14) controlled by the barrages in the main river and this result is in agreement with the findings of Noh et al. (2015) that demonstrated the increase of those variables in the regulated water environment in lab experiments. Consequentially, the barrages have significantly contributed to the enhanced role of water temperature in the main river as the key variable.

Moving forward, the increasing trend of temperature as a result of climate change will likely make a bigger impact on this lentic environment in the main river by enhancing the heating process as well as causing lags in the cooling and heating processes. These changes in the thermal system in the river will eventually create more variability in diatoms along with the introduction of new or exotic species amid climate change. Similarly, Lee et al. (2021) warn that the use of the mTDI is going to be vulnerable to the introduction of exotic species. Therefore, the current use of the index, which merely captures a snapshot of the biological or ecological state of rivers through one or two-time visits per year will become less accurate for an understanding of water quality and river health. This implies that the responses of diatoms to water quality and seasonality should be considered together in temporal aspects for more accurate water quality and river health assessment.

Regarding the responses of diatoms to seasonal changes, it is worth looking at three studies (Snell et al., 2014; Zou et al., 2018; Snell et al., 2019) that have turned their focus into investigating successive seasonal changes of diatom assemblages. Zou et al. (2018) demonstrated using two years of monthly diatom samples that water temperature was the main driver for seasonal changes of several diatom species, not the trophic state in Yunlong Lake in China. For instance, species like *Aulacoseira granulata* var. *angustissima*, *Achnanthidium catenatum*, *Aulacoseira ambigua*, *Discostella asterocostata*, and *Lindavia balatonis* exhibited obvious seasonal patterns in relation to water temperature. Although this study was carried out in a lake, it empirically demonstrates the presence of the seasonal succession of diatom assemblages in freshwater and the important role of water temperature on the seasonal succession. In two headwater streams in the U.K, Snell et al. (2014) confirmed the strong similarities in the overall structural and functional benthic ecosystem changes using the two-years of monthly diatom monitoring data despite the dynamic nature of the physical environment. Furthermore, Snell et al. (2019) constructed a robust and recurring seasonal cycle of diatoms in three low order streams in England using a ratio of certain species. In their study, the taxa identified in three catchments were slightly different, but ratios of them revealed repeated seasonal patterns during six years, which coincides with the result of the ecological health assessment; a ratio of *Achnanthidium minutissimum* to *Amphora pediculus* was identified for Pow Beck and Newby Beck catchment, while a ratio of *Gomphonema parvulum* to *Navicula lanceolata* was found to be related with ecological health assessment for Thackthwaite Beck catchment. Snell et al. (2019) concluded that this seasonal trend provides a baseline to identify the impact of land-use decisions on the response of diatoms at the catchment scale. These two examples (Zou et al., 2018; Snell et al., 2019) suggest the possibility that the seasonal changes of diatoms can be used as groundwork for water quality assessment.

This project has also found a strong sign of recurring seasonality in diatom assemblages in some sites. For instance, T11a is the prime example that has seen a regular switch in NMDS space between spring and autumn (see T11a in Figure 5.22). This regular movement in NMDS space is then translated into alternating occurrences of diatom groups from G1 to G3b in the distribution of diatom groups (Figure 6.3). Furthermore, those taxa identified in the study of Snell et al. (2019) are common species frequently found in the Nakdong river and its tributaries. It is also reported that several species like *Achnanthes minutissima*, *Achnanthes convergens*, *Navicula subminuscula*, *Navicula minima*, and *Nitzschia inconspicua* have seasonal differences in abundance in tributaries NS, YG, and WE (Choi et al., 2019). Therefore, these signs from this study and the findings from Choi et al. (2019) imply the possibility that a seasonally-recurring baseline of diatoms would exist in the Nakdong river and its tributaries. However, it may be difficult to detect a seasonal pattern of diatoms in the main river as the change of diatom groups between spring and autumn is less obvious in the main river (Figure 6.3) due to the effect of the barrages as well as the complexity in high order streams.

7.3.3 A holistic approach to river health assessment

For diatoms to be better used in river health assessment, a holistic approach is needed. To date, most water quality research using diatoms is heavily focused on the application of a diatom index on a short time scale (e.g. usually on a year scale, see Table A.1), while the species composition of diatom assemblages has been left ignored. This project demonstrates that the simple application of the mTDI or diversity indices is not sufficient to deliver a broad picture of river health on a long-term scale (Figure 5.3 and 5.4).

Instead, the findings of this project provide the whole story of river health in the Nakdong river and its tributaries based on the close connection between water quality and epilithic diatoms during 2009–2018 and these results enhance our understanding of river health in the catchment. Therefore, a holistic understanding of river health using diatoms can be achieved when five conditions are considered together: 1) the species composition of diatom assemblages, 2) relationships between environmental variables and diatoms, 3) multiple years of diatom sampling data at an inter-annual scale, 4) multiple times of diatom sampling data at an intra-annual scale, 5) a longitudinal perspective at a spatial aspect.

In this study, the examination of the GOVD dataset using the species composition reveals dynamic responses of diatom assemblages to seasonality and environmental stressors such as the construction of the barrages, and the hydrological drought in the results of NMDS analysis and Community Trajectory Analysis (Figure 5.21–5.22, and Figure 5.24). Also, the classification of diatom groups based on similarity of structure provides a basis for river health assessment (Figure 6.3), which is understood based on the close relationships between environmental variables (Figure 6.5) and diatoms and the selected indicator species (Table 6.1). For instance, the worsening trends of water quality in the up- and midstream of the main river (Figure 3.20 and 3.22) are directly translated into the dominance of G3b in the upstream of the main river with the frequent occurrence of G3a in diatom groups by replacing G2. These two dominant subgroups are characterised by the highest organic pollution and Chla concentration compared to other groups. Furthermore, these temporal and spatial changes in diatom groups are only detectable at intra- and inter-annual scales from a longitudinal perspective and the current study is the first to detect these spatio-temporal changes of diatom groups in the main river as well as the first in the world to my knowledge.

These five conditions for river health assessment are not difficult to be met as many countries have a pile of multiple years of diatom monitoring datasets. Passy (2007a) also stresses the importance of temporal and spatial variables, which can be easily obtained and the need to integrate them to utilise diatoms in water quality and river health monitoring. Ultimately, the accomplishment of understanding holistic river health at the catchment scale will help local government and environmental engineers determine or establish the direction in river health management planning at an administrative level.

7.4 Suggestions for the better use of diatoms in river health assessment

Based on the findings of this research and discussion above, I propose three suggestions for diatoms to be better used in river health assessment. First, the widespread sampling practice of 'one visit per site' during the designated period (usually spring and autumn) in river health assessment is strongly discouraged and sampling should be taken once a month per site with a minimum of two visits per site during the period. In the field, temporal variables such as time, day, month and year should be noted and utilised instead of simply using the notion of spring or autumn. Also, diatom sampling should be strictly avoided at a time of survey when it is unusually cold or hot compared to the average state of climate to minimise unintended variability in diatom assemblages driven by temperature. In the literature, the recommendation for diatom sampling is only made regarding hydrological conditions (Kelly et al., 1998), whilst the impact of temperature on diatom assemblages is overlooked despite the study of Round (1991) acknowledging seasonal occurrences of diatoms in rivers. However, changes in water temperature can cause more variability in diatom assemblages and should be controlled in diatom sampling by avoiding sampling in unusual climate conditions.

Second, it is strongly required to switch the river health assessment method to monitoring the seasonal cycle of diatoms at a catchment scale from the current monitoring programme which relies on one or a few samples a year. In this way, we can move away from merely capturing a snapshot of the ecological status in specific sites in rivers, and progress towards grasping the temporal evolution of river health assessment at the catchment scale as a unit (e.g. the EU Water-Framework Directive). As a result, it will enhance our understanding of the catchment by looking at the response of diatom assemblages in ecosystems to any anthropogenic activities as well as climatic variables. This understanding will provide valuable insight into the response of the catchment and help local governments or decision-makers establish management plans at a catchment scale in successive temporal perspectives. To implement this idea, an appropriate catchment size in which the response of diatoms to environmental stressors is identifiable as a unit and the location of monitoring sites needs to be determined at the catchment level. For the construction of a robust seasonal cycle, monthly sampling is generally required based on administrative cost and effort, and the results of previous studies (Snell et al., 2014; Zou et al., 2018; Snell et al., 2019).

Finally, it is encouraged to conduct periodic river health research on the catchment scale at a regular interval of three to five years to provide a comprehensive understanding of river health. This project, covering the ten years of diatom records, demonstrates that the response of diatoms following the construction of the barrages is visible at the inter-annual scale, which enhances our understanding of the impact of the barrages on the ecosystem. Therefore, it is beneficial for local governments to assess the effectiveness of their river health management plans and develop them in the right direction on a long-term scale.

CHAPTER 8

Conclusion

8.1 Implications of the research

This project has examined the changes in water quality and diatom assemblages from 2009 to 2018 in the Nakdong catchment and attempted to understand their relationships with climatic variables, hydrological variables, and the construction of the barrages that took place between 2010–2012. This project and its findings have implications in three aspects.

First, amid the strong tendency in water quality assessment where epilithic diatoms are linked with water quality on a short-term scale, this research is meaningful as the first attempt to comprehend spatio-temporal changes in water quality and diatom assemblages on ten years of time scale at catchment scale. In particular, the findings are effective in demonstrating the responses of diatom assemblages to climatic variables, hydrological variables, water quality, and the construction of the barrages in the main river. In these findings, diatoms are central to the link between those changes. Therefore, these findings substantially strengthen the status of diatoms as indicators to detect environmental changes in rivers and ensure that diatoms can continue to be utilised in the future.

Secondly, the findings of this project demonstrate the impact of the barrages on the hydrological system, water quality, and diatom assemblages and will be useful as a prime example for other countries across the world who may consider building barrages or dams in rivers in the future. The findings do not necessarily mean that such construction should be prohibited but show that there needs to be sufficient consideration about what environmental impacts would be expected to happen and what measures should be taken to mitigate the impacts on riverine ecosystems. In addition, this research discovers the different responses of diatoms to the hydrological changes; the barrages altered the flow regime, whilst the weirs were insufficient to change the flow entirely. This suggests that the current interchangeable use of both terms in the literature may lead to confusion and there is a need to differentiate the terms 'barrages' and 'weirs' in future research.

Third, this project is the first research that applies a novel framework of Community Trajectory Analysis (De Cáceres et al., 2019) to a diatom dataset on top of the use of conventional ordination methods. The sequence of changes of diatoms over time, described as geometric vectors, demonstrates its utility in explaining the temporal change of the ecological communities over time. Hence, the use of this method will facilitate comparisons of diatom assemblages in ecological studies.

8.2 Implications of the research for South Korea

This project and its findings have significant meanings for environmental engineers and researchers in South Korea. First, this project is meaningful as the first research that has examined the impact of the barrages based on the connection between water quality and diatom assemblages in the Nakdong river and its tributaries using ten years of data. Thus, the findings of this research can be used as a basis for the South Korean government and environmental managers to re-assess the impact of the barrages on the ecosystems. The results can be also utilised to raise public awareness of the impact of the barrages in the rivers.

Second, three suggestions this project proposed can be seamlessly adapted to the existing river health monitoring programme in South Korea. In particular, the examination of successive seasonal changes of diatoms can be tested by making the best use of some of the current monitoring sites. Based on the results, a consecutive diatom monitoring scheme can be put in place with a catchment-specific approach to see the response of diatoms to land use activities at the catchment scale. This way, environmental engineers can control the impact of non-point pollution sources on river health and establish management plans within the catchment. The periodic river health assessment can easily be brought in within the current river health monitoring programme as it entirely relies on the accumulated results of the annual diatom monitoring programme.

Finally, this research will encourage researchers in South Korea to take a holistic approach to understanding riverine ecology, moving away from the emphasis on water quality. This may also inspire them to take a similar approach with other types of river health monitoring indicators such as benthic macroinvertebrates and fish.

8.3 Directions for further research

The findings of this research can be bolstered further with the following research ideas from a local scale to an international scale.

1. At the local scale, the impact of the barrages on diatom assemblages in the main river can be assessed further regarding the relative location of sites to the barrages and hydrological features such as current velocity. In addition, this project can be scaled down to a tributary level in the Nakdong catchment where the responses of diatoms in the tributaries can be examined in detail in relation to changes in land use and hydrology. By doing this, we will be able to better understand the time-series observations for the tributaries. For example, tributary NS has the Yeongju Dam upstream which started operating in the summer of 2017 and caused significant changes in channel morphology and flow.

2. At the regional scale, it is necessary to keep looking at the responses of diatom assemblages in the main river following the decision to leave the gates open at five barrages since 2018 summer (partial opening at B1, B5, B6 and B8; full opening at B7; no change at B2–B4).

3. At the national scale, there are three other major rivers in other parts of South Korea that have barrages built during the same period as part of the FMRRP in 2009: the Han River, the Geum River, and the Youngsan River. Thus, it is worth examining the responses of diatoms in those river networks

to assess the impact of the barrages on water quality and diatom assemblages in the rivers.

4. At the international scale, more evidence on the response of diatoms to hydrological changes such as hydrological drought and the construction and/or removal of artificial structures in rivers should be gathered to enhance our understanding of diatoms to those changes.

By gathering evidence at various scales and comparing these with others, the impacts of barrages on the riverine ecosystems can be better understood in depth. These understandings ultimately will provide a baseline for river management planning, which can sustain humans and ecosystems.

Appendix A

Selected literature for river health assessment

Table A.1: Selected literature for river health assessment using diatoms in rivers

Author (Year)	Study Site (River)	Number of Sampling sites	Sampling frequency	Study Time Scale	Research Summary	Topic
Molloy, 1992	The U.S.	18	4	1 month	Longitudinal gradient of diatoms (diatom morphological guilds)	Pollution
Lobo et al., 1995	Japan	52	1	Once	Response of diatoms to water pollution in Tokyo, Japan	Pollution
Rott et al., 1998	Canada	10	5	5 months	Organic pollution and eutrophication by means of diatoms	Pollution
Newall and Walsh, 2005	Australia	16	2	1 year	Response of diatom assemblages to urbanization influences	Land use
Camargo and Jiménez, 2007	Spain	4	1	Once	Fish farm pollution; response between diatoms and macrophytes	Land use
Walsh and Wepener, 2009	South Africa	7	1	Once	The influence of land use on water quality and diatom community structures	Land use
Tan et al., 2014	China	29	1	Once	Spatial pattern of benthic diatoms and water quality assessment using diatom indices	Pollution
Chen et al., 2016	China	29	1	Once	Diatoms as indicator for urban stream conditions	Land use
Zou et al., 2018	China	1	24	2 year	Seasonal diatom variability in lake using sediment trap records	Seasonality
Munn et al., 2018	The U.S.	98	1	Once	Assessing the influence of multiple stressors on stream diatom metrics	Various factors
Snell et al., 2014	The U.K.	2	25	2 years	Duration of diatom community representivity and response periods in headwater streams	Seasonality
Snell et al., 2019	The U.K.	3	72	6 years	Seasonal change in stream diatom assemblages	Seasonality
Krajenbrink et al., 2019	The U.K.	77	2	4 years	Diatoms as indicator of the effects of river impoundment at multiple spatial scales	Impoundment
Yoon et al., 2010	Korea (Wonju Stream in Han River Watershed)	14	4	1.5 year	Effects of pollution from urban river to diatom assemblages using TDI	Pollution
Hwang et al., 2011	Korea (All rivers and streams)	720	1	Once	Distribution of benthic diatoms in relation to environmental variables	Various factors
Kim et al., 2012	Korea (North Han River)	9	6	3 years	Biological water quality assessment using DAIpo and TDI	Pollution
Kim et al., 2013	Korea (24 small rivers and streams in south and west region)	38	2	Once	Comparisons of water quality and diatoms between regulated streams and unregulated streams	Pollution
Kim and Lee, 2017	Korea (Hangae and Buk Stream)	20	4	1 year	Altitudinal distribution of diatoms in mountainous streams	Altitude
Kim et al., 2017	Korea (Naeseong Stream)	15	3	1 year	Biological water quality assessment using DAIpo and TDI	Pollution
Pandey et al., 2018	Korea (Han, Geum, Youngsan, and Nakdong)	16	1	Once	Four major rivers flowing near industrial complexes and municipal effluents	Pollution
Choi et al., 2019	Korea (Young River, Naeseong Stream, We Stream)	14	3	1 year	Biological water quality assessment using DAIpo and TDI	Pollution
Lee et al., 2021	Korea (Nakdong River)	16	4	1 year	Response of diatoms to weir construction (only dealt with diatom samples in 2014)	Pollution

Appendix B

Monthly discharge in the tributaries

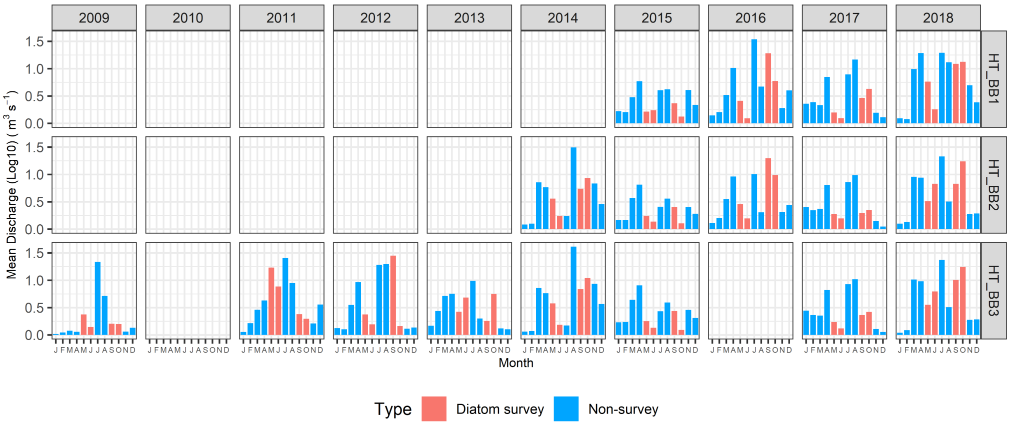


Figure B.1: Monthly discharge at the catchment BB. An empty plot indicates no data. Summer peak discharge in 2015 was significantly low due to little precipitation. BB1 is a flow station upstream of the Imha Dam (D_IMHA) and BB2 and BB3 are from another tributary below T1. These two sites do not directly affect T1, nevertheless, they were included to see the spatio-temporal pattern of precipitation and discharge in the catchment. The red colour represents diatom survey months in the national programme, while the blue colour represents non-survey months.

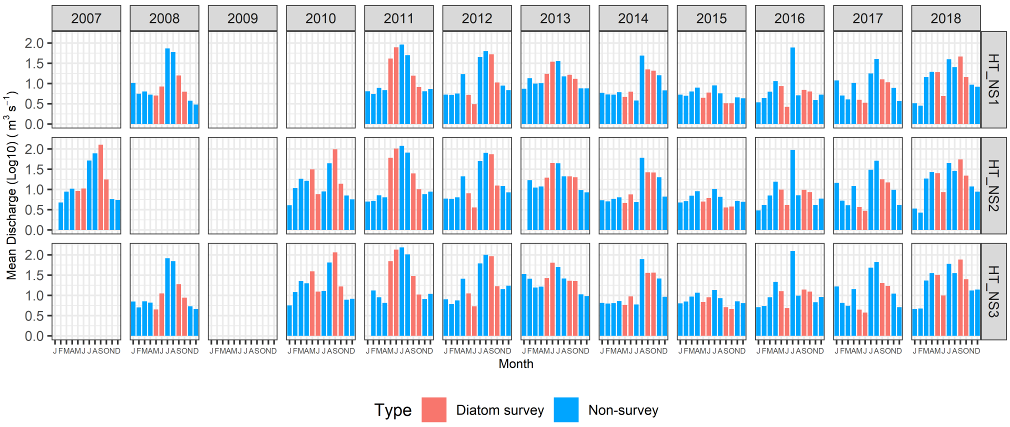


Figure B.2: Monthly discharge at the catchment NS. An empty plot indicates no data. Summer peak discharge in 2015 was significantly low due to meteorological drought. The Yeongju Dam (D_YENJ), located upstream of those three flow gauging stations, started operating in July 2017. The red colour represents diatom survey months in the national programme, while the blue colour represents non-survey months.

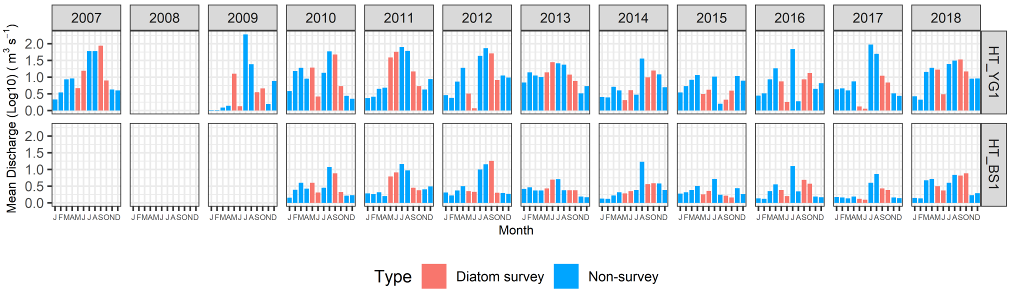


Figure B.3: Monthly discharge at the catchments YG and BS. An empty plot indicates no data. Due to meteorological drought in 2015, monthly mean discharge in summer was considerably low in catchment YG. Overall, tributary BS has a relatively low mean discharge compared to YG. Summer peaks in 2013 and 2015 were lower than the average level in both tributaries. The red colour represents diatom survey months in the national programme, while the blue colour represents non-survey months.

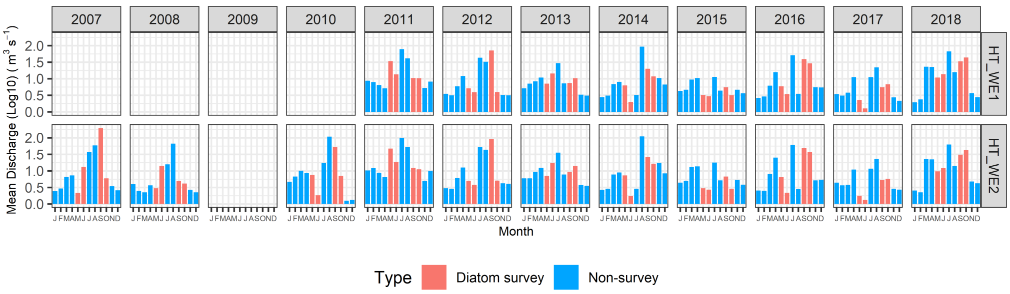


Figure B.4: Monthly discharge at the catchment WE. An empty plot indicates no data. Tributary WE had lower summer peaks in 2013 and 2015 because of little precipitation. The red colour represents diatom survey months in the national programme, while the blue colour represents non-survey months.

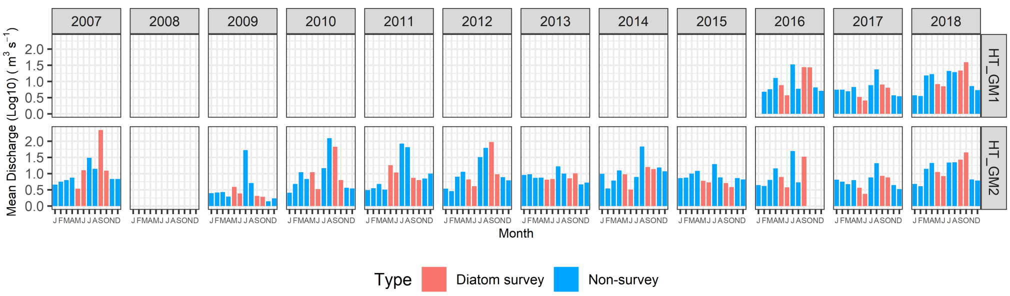


Figure B.5: Monthly discharge at the catchment GM. An empty plot indicates no data. The disappearance of the summer peak in 2013 and 2015 is related to the dry periods. The red colour represents diatom survey months in the national programme, while the blue colour represents non-survey months.

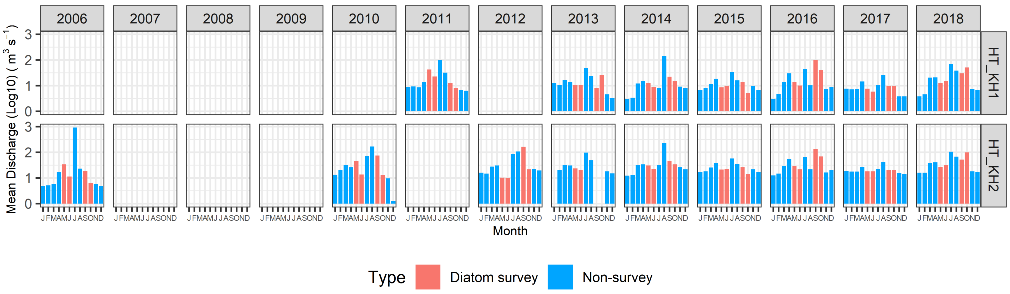


Figure B.6: Monthly discharge at the catchment KH. An empty plot indicates no data. Monthly discharge in tributary KH is evenly distributed throughout the season compared to other tributaries. In tributary KH, discharge in 2017 was lower than in 2015, but the difference is not significant. During the two years, the summer peak was lower than the average level because of little precipitation. The red colour represents diatom survey months in the national programme, while the blue colour represents non-survey months.

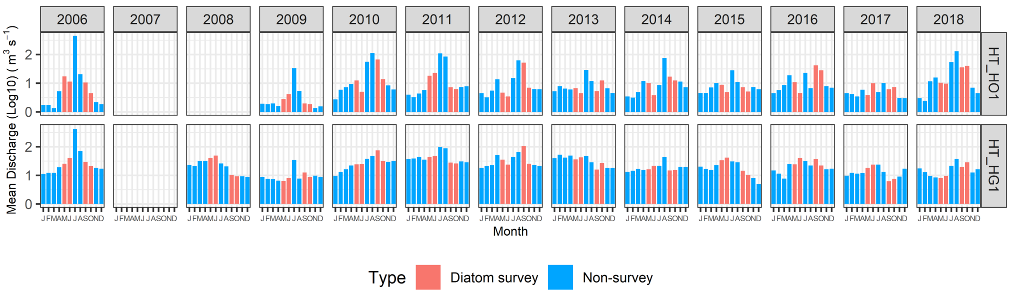


Figure B.7: Monthly discharge at the catchments HO and HG. An empty plot indicates no data. Tributary HO was the driest in 2009, followed by 2017. Tributary HG has a fairly even distribution of monthly discharge due to the Hapcheon Dam (D_HAPC) controlling flow. The red colour represents diatom survey months in the national programme, while the blue colour represents non-survey months.

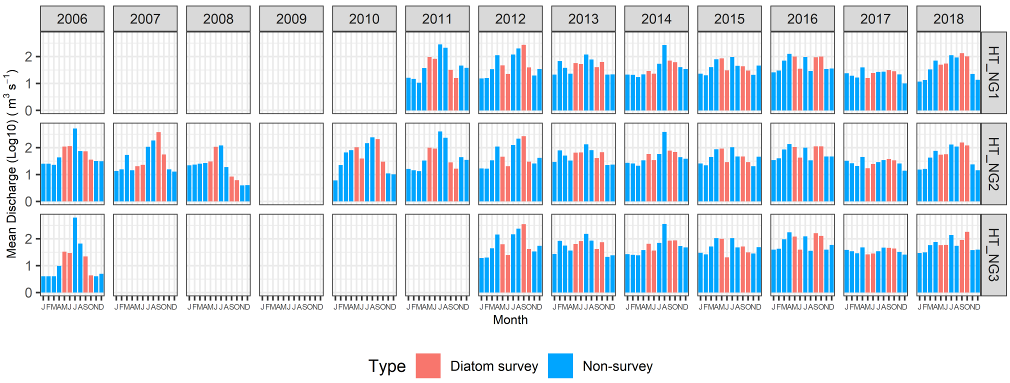


Figure B.8: Monthly discharge at the catchment NG. An empty plot indicates no data. Tributary NG, controlled by the Namgang Dam (D_NAMG) upstream, shows a stable level of monthly discharge throughout the year despite the two meteorological droughts in 2015 and 2017. Although discharge in 2017 had no clear summer peak, its impact would not be as great as in other small tributaries. The red colour represents diatom survey months in the national programme, while the blue colour represents non-survey months.

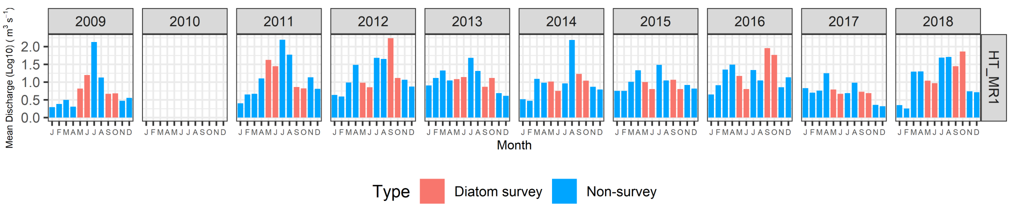


Figure B.9: Monthly discharge at the catchment MR. An empty plot indicates no data. Due to the meteorological droughts in 2015 and 2017, the summer peak in discharge did not form at all. Except for that, there is no significant change to the pattern of monthly discharge over time. The red colour represents diatom survey months in the national programme, while the blue colour represents non-survey months.

Appendix C

Mean changes in water quality

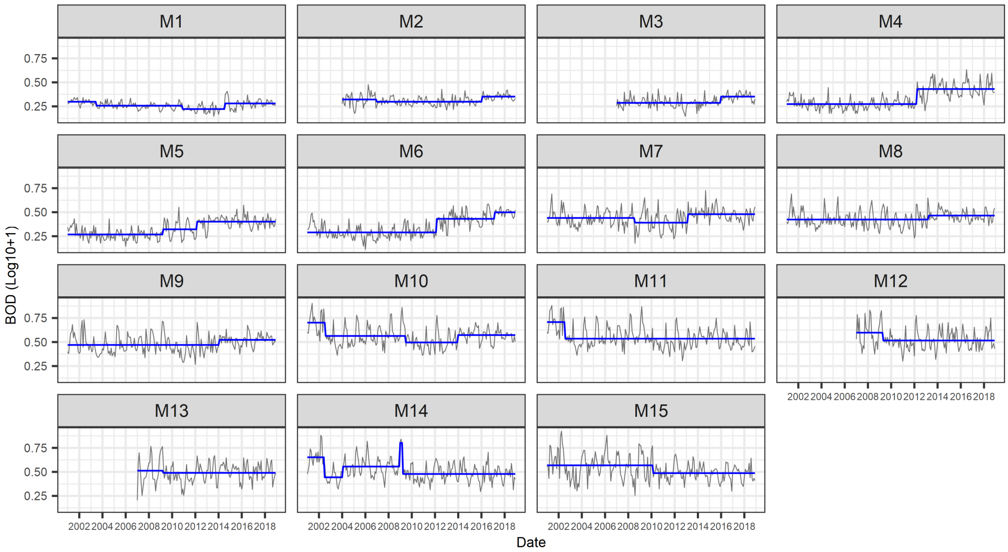


Figure C.1: Mean changes of monthly observed biological oxygen demand (BOD) in the main river from 2001 to 2018. The grey lines are monthly observations and the blue solid lines are mean values.

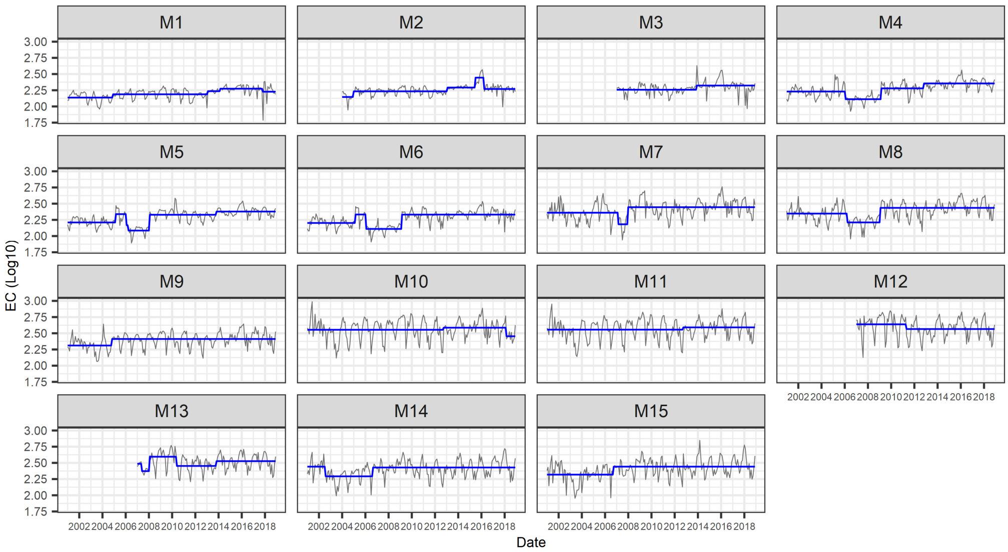


Figure C.2: Mean changes of monthly observed electrical conductivity (EC) in the main river from 2001 to 2018. The grey lines are monthly observations and the blue solid lines are mean values.

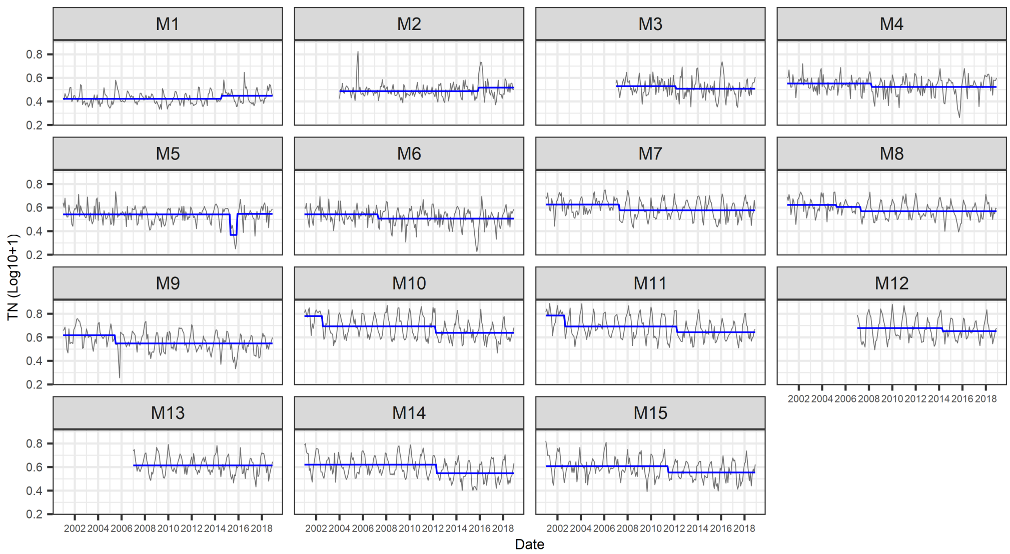


Figure C.3: Mean changes of monthly observed total nitrogen (TN) in the main river from 2001 to 2018. The grey lines are monthly observations and the blue solid lines are mean values.

Appendix D

The results of analyses on the GOVD dataset

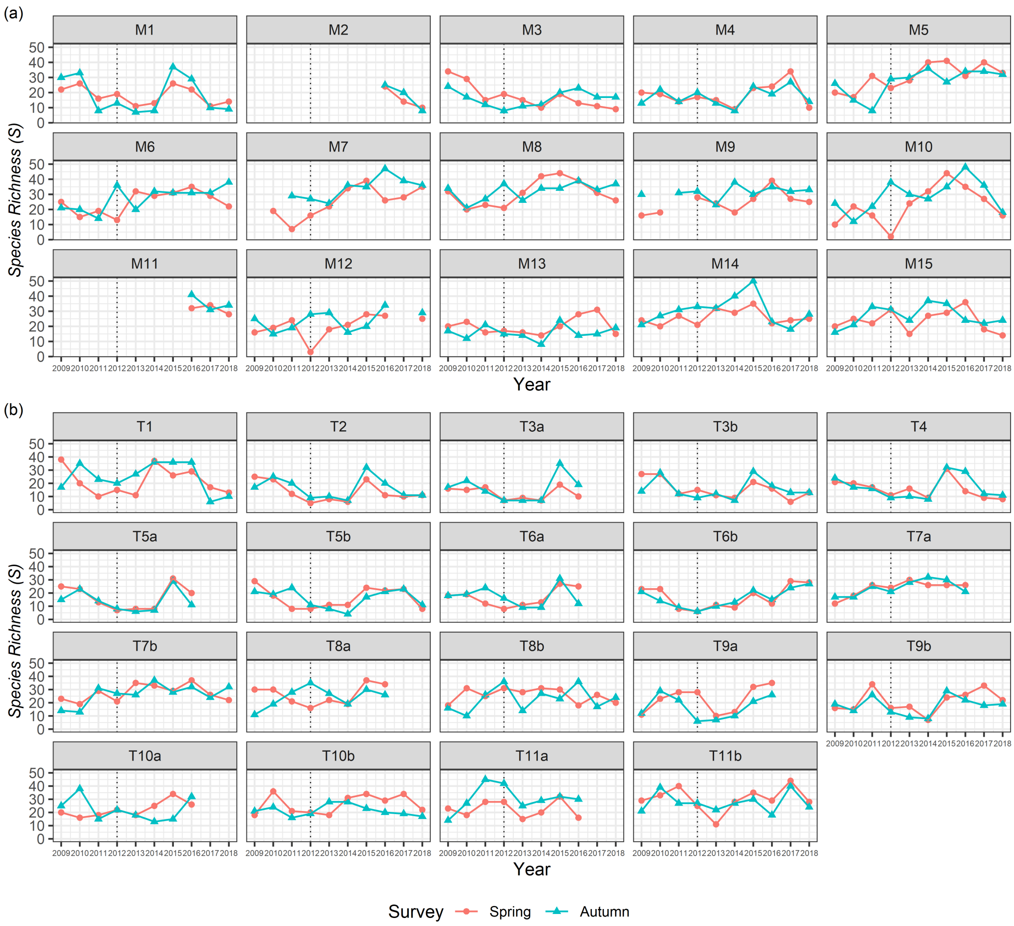


Figure D.1: Distribution of Species Richness (S) from 2009 to 2018 by sites and surveys. (a) The main river, (b) The tributaries. Vertical dashed lines indicate the year when the construction of the barrages was complete.

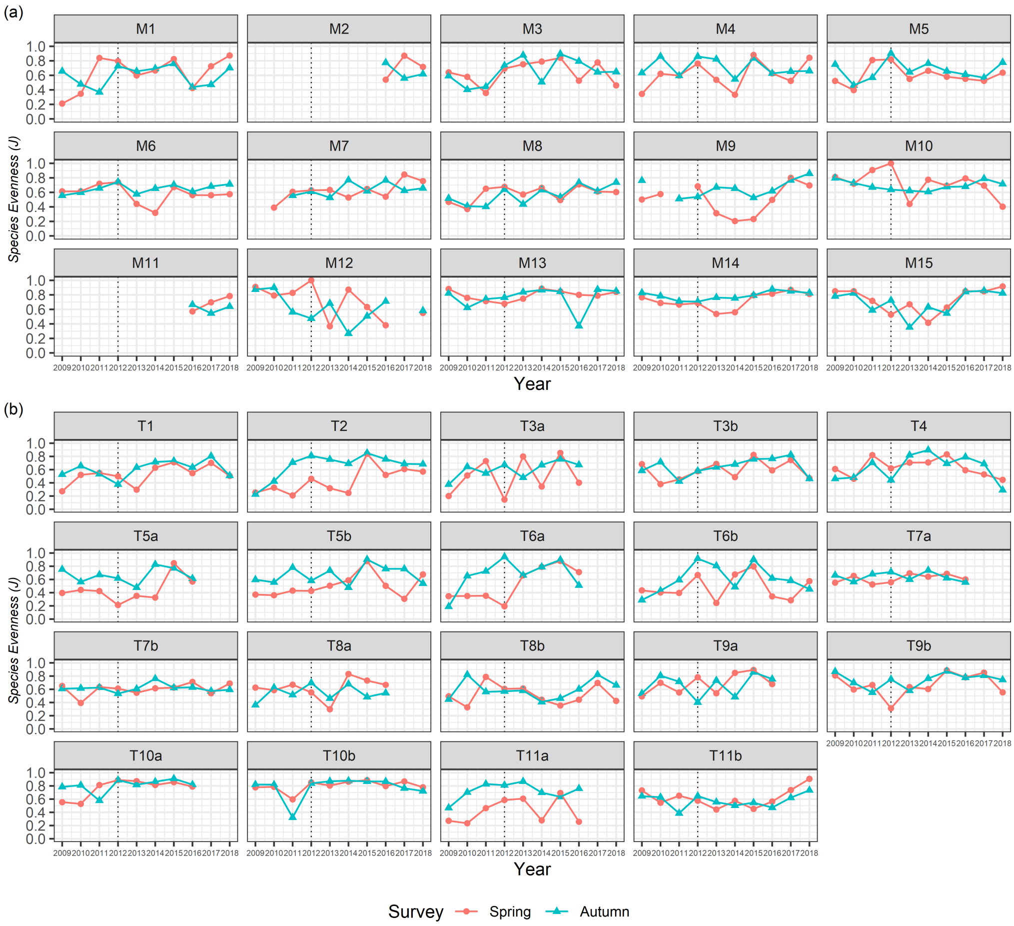


Figure D.2: Distribution of Species Evenness (J) from 2009 to 2018 by sites and surveys. (a) The main river, (b) The tributaries. Vertical dashed lines indicate the year when the construction of the barrages was complete.

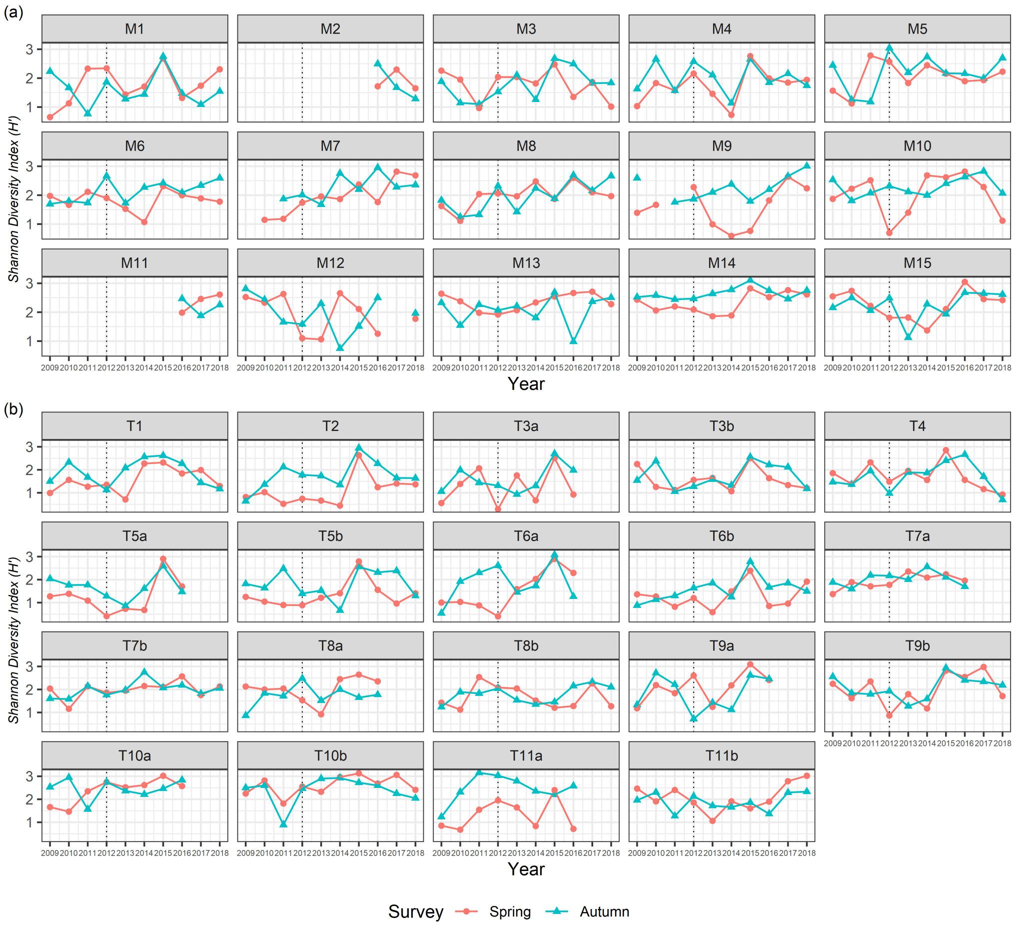


Figure D.3: Distribution of Shannon Diversity Index (H') from 2009 to 2018 by sites and surveys. (a) The main river, (b) The tributaries. Vertical dashed lines indicate the year when the construction of the barrages was complete.

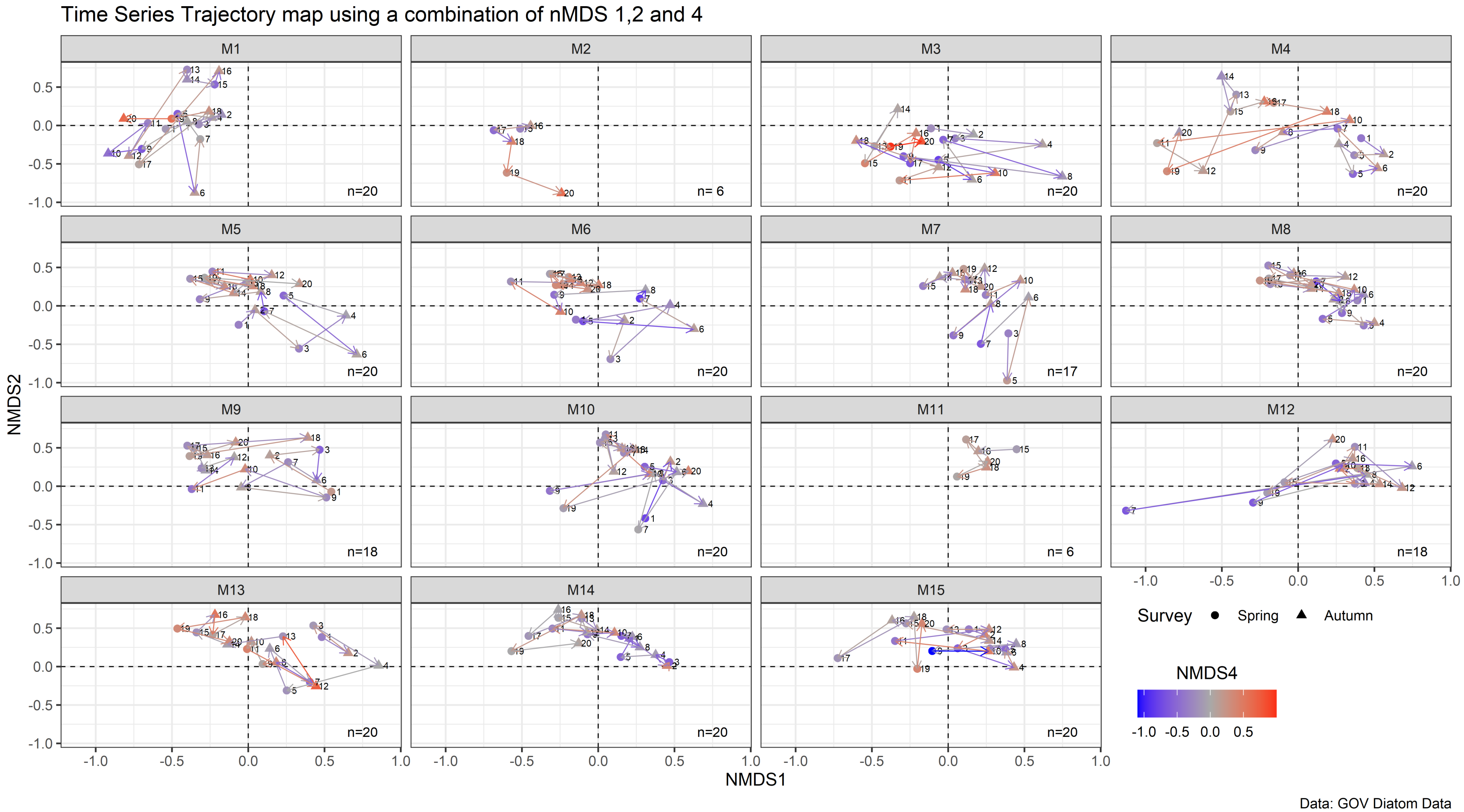


Figure D.4: Time-series trajectories of diatom assemblages from M1 to M15 in the main river by survey using a combination of NMDS1, 2, and 4. The number of observations (n) varies by site and dashed vertical and horizontal lines are arbitrarily drawn for ease of reading.

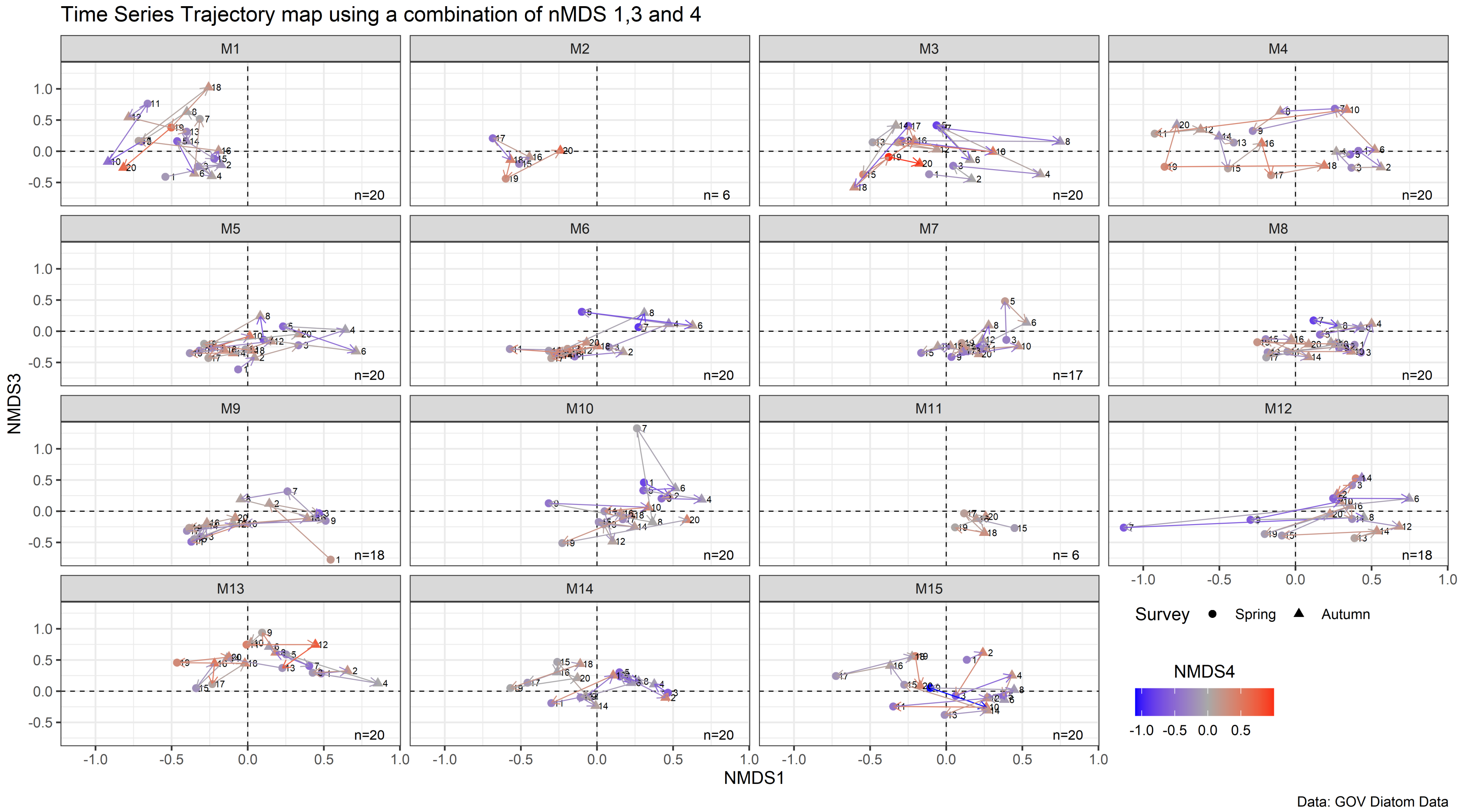


Figure D.5: Time-series trajectories of diatom assemblages from M1 to M15 in the main river by survey using a combination of NMDS1, 3, and 4. The number of observations (n) varies by site and dashed vertical and horizontal lines are arbitrarily drawn for ease of reading.

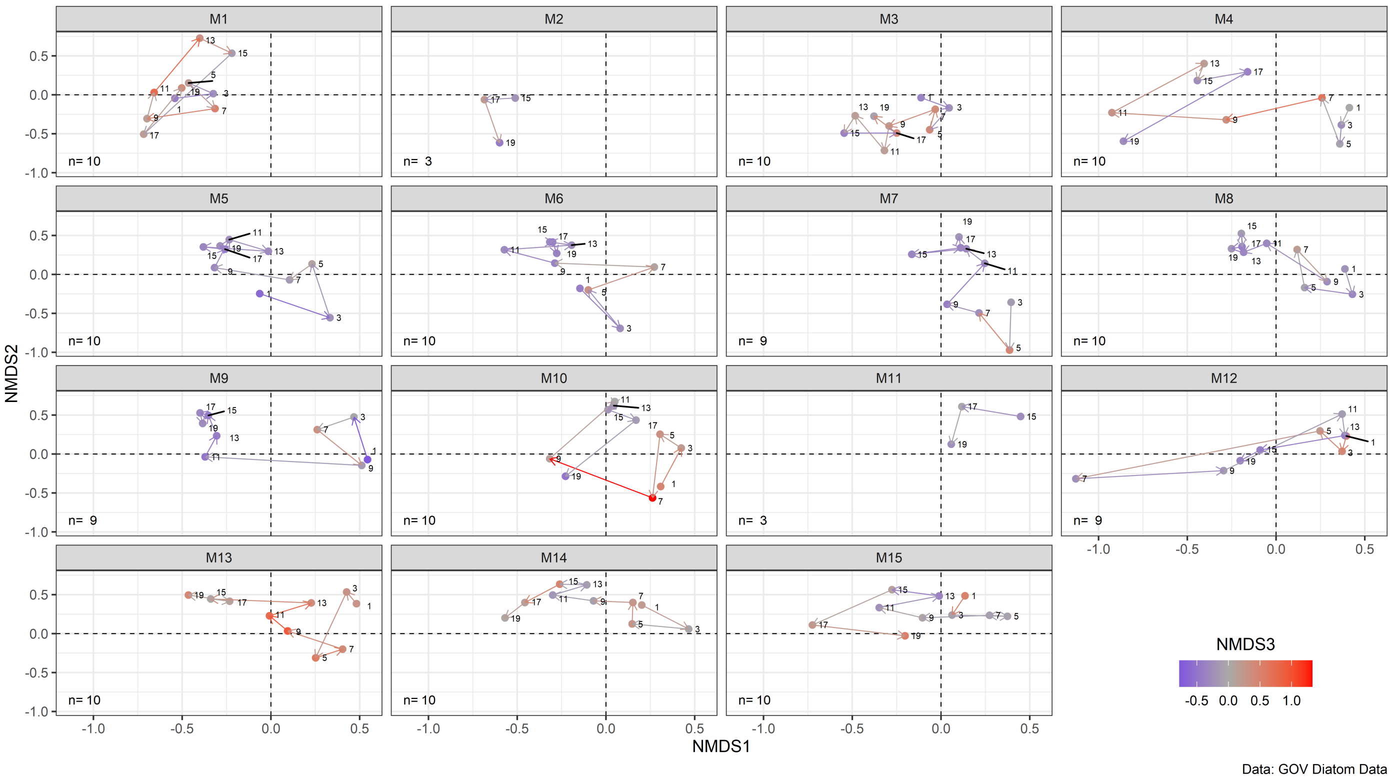


Figure D.6: Time-series trajectories of diatom assemblages by site in the main river for spring. Stage is numbered in chronological order of survey. Odd number is sampled in spring, while even number is in autumn; 1-2009 Spring, 2-2009 Autumn, 3-2010 Spring, 4-2010 Autumn, 5-2011 Spring, 6-2011 Autumn, 7-2012 Spring, 8-2012 Autumn, 9-2013 Spring, 10-2013 Autumn, 11-2014 Spring, 12-2014 Autumn, 13-2015 Spring, 14-2015 Autumn, 15-2016 Spring, 16-2016 Autumn, 17-2017 Spring, 18-2017 Autumn, 19-2018 Spring, 20-2018 Autumn. The number of observations (n) varies by site and dashed vertical and horizontal lines are arbitrarily drawn for ease of reading.

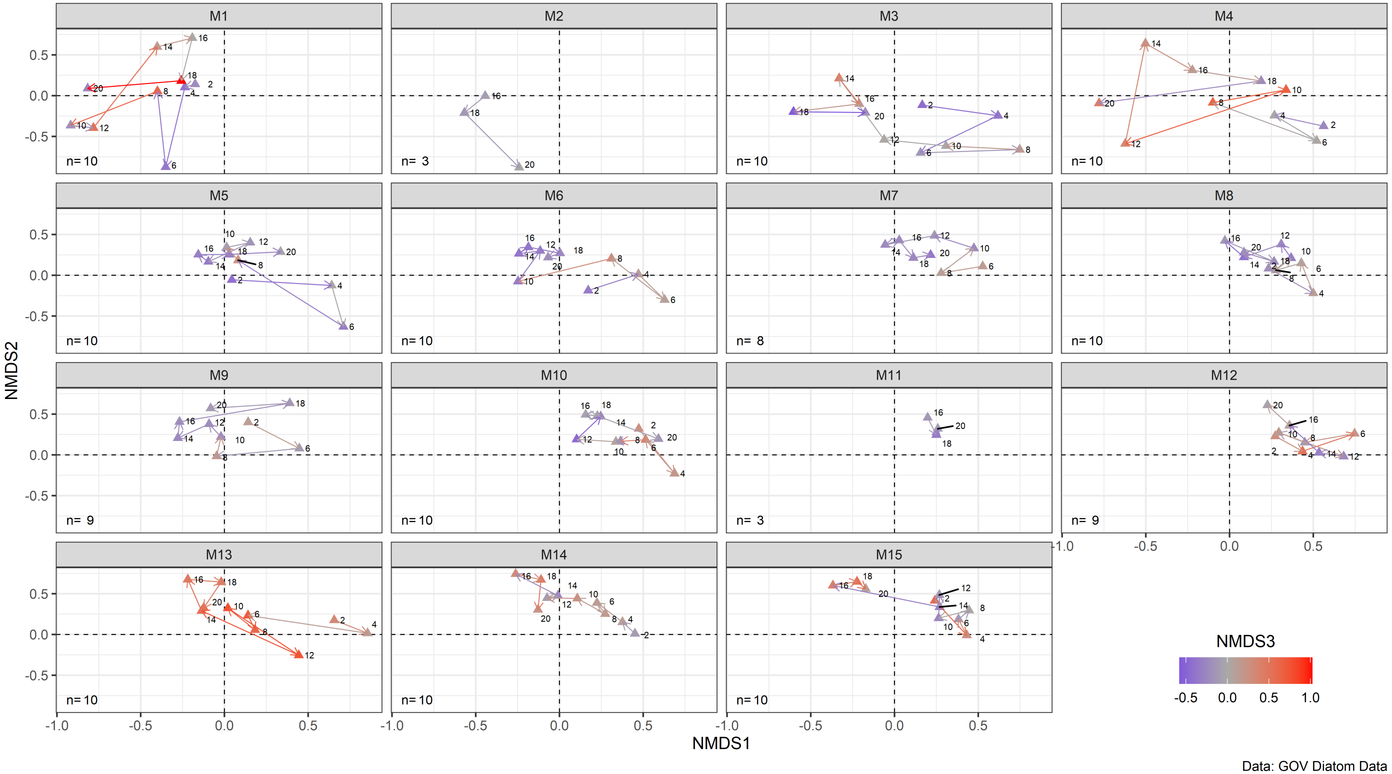


Figure D.7: Time-series trajectories of diatom assemblages by site in the main river for autumn. Stage is numbered in chronological order of survey. Odd number is sampled in spring, while even number is in autumn; 1-2009 Spring, 2-2009 Autumn, 3-2010 Spring, 4-2010 Autumn, 5-2011 Spring, 6-2011 Autumn, 7-2012 Spring, 8-2012 Autumn, 9-2013 Spring, 10-2013 Autumn, 11-2014 Spring, 12-2014 Autumn, 13-2015 Spring, 14-2015 Autumn, 15-2016 Spring, 16-2016 Autumn, 17-2017 Spring, 18-2017 Autumn, 19-2018 Spring, 20-2018 Autumn. The number of observations (n) varies by site and dashed vertical and horizontal lines are arbitrarily drawn for ease of reading.

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