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Impact of Conservation in the Futian Mangrove National Nature Reserve on Water Quality in the Last Twenty Years

Jin Luo [1,2], Qiming Huang [1,*], Hongsheng Zhang [3], Yanhua Xu [1], Xiaofang Zu [1] and Bin Song [1]

1   School of Geography and Environment, Jiangxi Normal University, Nanchang 330022, China; luojin@jxnu.edu.cn (J.L.); yanhuaxu@cnu.edu.cn (Y.X.); zxf@jxnu.edu.cn (X.Z.); 202040100124@jxnu.edu.cn (B.S.)
2   Key Laboratory of Poyang Lake Wetland and Watershed Research, Ministry of Education, Jiangxi Normal University, Nanchang 330022, China
3   Department of Geography, The University of Hong Kong, Pokfulam, Hong Kong, China; zhanghs@hku.hk
*   Correspondence: qmhuanggis@jxnu.edu.cn

Abstract: Mangroves play a crucial role in improving the water quality of mangrove wetlands. However, current research faces challenges, such as the difficulty in quantifying the impact of mangroves on water quality and the unclear pathways of influence. This study utilized remote sensing imagery to investigate the long-term changes in mangrove forests in the Futian Mangrove National Nature Reserve and constructed a water quality index based on water quality data. Finally, structural equation modeling was employed to explore the pathways of influence and quantify the impact effects of mangroves, climate, and water quality. The study findings revealed several key points: (1) The mangrove forests in the Futian Mangrove National Nature Reserve exhibited a trend of expansion towards the ocean during this period. (2) The seasonal and annual characteristics of water quality in Shenzhen Bay indicated a significant improvement in water quality from 2000 to 2020. (3) Mangroves have significant direct and indirect impacts on water quality, which are more pronounced than the effects of climate factors. These findings not only offer insights for the environmental management and conservation of Shenzhen Bay but also provide support for future comprehensive studies on the response relationships between the morphology, species, and physiological characteristics of mangroves and water quality.

Keywords: mangrove; improvement of water quality; structural equation modeling; climate impacts; Shenzhen Bay

1. Introduction

Mangrove forests, found in the coastal intertidal zone of tropical and subtropical regions, are a unique vegetation complex that provides essential ecological services and economic functions, including water purification. According to Edward B. Barbier [1], mangrove ecosystems play a crucial role in carbon storage, biodiversity maintenance, coastline protection, and water purification [2–4]. It is noteworthy that mangrove forests play a crucial role not only in stabilizing ecosystems and preserving biodiversity [5,6] but also in positively impacting the coastal environment and human society in various ways [7,8].

Research data show that the total area of mangrove forests globally is approximately 147,000 km^2, which represents only 0.1% of the global land area [9]. Despite their small proportion of the global land mass, mangrove forests are responsible for 5% of total global carbon fixation [10], making them one of the most carbon-intensive ecosystems on land [11]. This is due to the strong carbon sequestration capacity of mangrove ecosystems. Additionally, mangroves and their ecosystems hold significant economic value, such as for aquaculture, food, and medicinal uses [12,13].

However, since the 1950s, the global mangrove area has been dramatically reduced and, in certain areas, has completely disappeared due to both natural and human activities [14–17]. Research results indicate that 35% of the global mangrove forests have disappeared and nearly 50% of the mangrove forests in China have vanished [18,19]. Therefore, mangrove conservation has become a major focus of research attention.

In 1985, China formally established the Futian Mangrove National Nature Reserve and implemented relevant regulations and protection measures. In 2015, the Guangdong Province of China established the Futian Mangrove National Nature Reserve Administration to enhance the supervision and management of mangrove forests in the reserve, as well as to conduct protective scientific research. Taking a further step, in 2022 the Chinese government announced the establishment of an International Mangrove Center in Shenzhen, Guangdong Province, to actively promote the conservation management and ecological restoration of mangrove wetlands.

Previous studies have widely agreed that the growth of mangroves has a positive impact on water quality, attributed to their unique ecosystem properties [20–22]. First, the extensive aquatic root systems of mangroves efficiently filter water and stabilize organic matter [23], reducing suspended solids and sediments and significantly improving water quality. Second, mangrove trees aid in water purification by absorbing harmful substances, such as heavy metals and organic pollutants, from both the benthic and suspended layers [24–26]. Additionally, the growth of mangroves helps to prevent soil erosion and reduce the entry of sediment into the water body, thereby slowing down the development of eutrophication [27–29].

Climate plays a critical role in the growth of mangrove forests and has a significant impact on water quality. Changes in temperature and precipitation patterns caused by climate change directly influence mangrove ecosystems [30–32], leading to shifts in distribution, growth rates, and nutrient cycling. Additionally, extreme events such as sea level rise and more frequent hurricanes can affect the dynamics and ecological processes of water bodies in mangrove areas. These changes can result in shifts in suspended solids, nutrients, and harmful substances in water, ultimately impacting water quality [33,34]. Therefore, gaining a comprehensive understanding of the combined impacts of climate change on mangrove forests and water quality is essential for developing effective strategies for ecological conservation and water quality management to adapt to changing environmental conditions.

Nonetheless, existing research still faces certain limitations in elucidating the mechanisms through which climate and mangroves influence water quality. Firstly, the intricacies of the climate system and mangrove ecosystems, involving numerous variables, pose challenges for conducting comprehensive and in-depth quantitative experimental studies [35]. Secondly, the interactions between mangrove ecosystems and water quality are intricate and complex, and a comprehensive understanding of the mechanisms underlying their impacts on water quality is still lacking [36]. Furthermore, there exists an indirect and progressive chain of impacts between climate change and mangrove water quality, making it challenging to explicitly quantify their direct effects. However, as the impacts of climate change on mangroves and water quality become increasingly apparent, delving into the role of mangroves in addressing climate change's effects on water quality becomes crucial not only for maintaining the health of marine and coastal ecosystems but also for providing vital support for environmental conservation and sustainable development.

To tackle these challenges, we extracted the mangrove area of the Futian Mangrove National Nature Reserve (FMNNR) from 2000 to 2020 using remote sensing images, enabling a comprehensive exploration of the growth dynamics of mangrove forests over an extensive period. Simultaneously, we utilized water quality data from Shenzhen Bay, sourced from the Environmental Protection Department of the Hong Kong Special Administrative Region of China, to construct a Water Quality Index (WQI) that characterizes the water quality status. This approach aimed to deepen our understanding of the evolving trends in water quality within the waters of Shenzhen Bay.

To conduct a more thorough examination of the factors influencing water quality, we incorporated climatic data spanning from 2000 to 2020. This dataset encompassed average monthly precipitation, average monthly air temperature, and interpolated monthly data derived from the fitting equation of the mangrove area. Through the integration of these climatic variables with water quality factors from the Shenzhen Bay dataset and the constructed WQI, we employed structural equation modeling. This analytical approach allowed us to delve into both the direct and indirect effects of mangroves and climate on water quality. By elucidating these intricate influence pathways, our study aims to offer deeper insights into the complex relationships between mangroves, climate, and water quality.

2. Materials and Methods

2.1. The Research Framework

In the present investigation, the gmap function from the dismo package in R v. 1.3-14. [37] was employed to acquire high-resolution remote sensing images. A total of six time periods were captured, with images from the years 2000 and 2004 obtained from the QuickBird satellite at a spatial resolution of 0.6m. However, for the years 2008, 2012, 2016, and 2020, images were sourced from the WorldView satellite series, exhibiting a spatial resolution range of 0.3–0.5 m.

In the initial step, from these high-resolution images, we derived the land use and its changes within the FMNNR via visual interpretation. Moreover, given the challenging assessment of the impact of water quality on mangrove vegetation growth and the coarse temporal resolution of images captured every four years, we employed Landsat 5 and Landsat 8 images. Using a random forest method for supervised classification, we aimed to complementarily extract the mangrove area for each year.

Next, we collected water quality data through online sources to construct a Water Quality Index (WQI), aiming to explore trends in water quality changes.

Finally, we combined climate data, encompassing precipitation data at a spatial resolution of $0.1° \times 0.1°$ and temperature data at a resolution of 1 km. Utilizing structural equation modeling, we aimed to quantify the effects of climate and mangroves on water quality and investigate the pathways of interaction among water quality influencing factors (Figure 1).
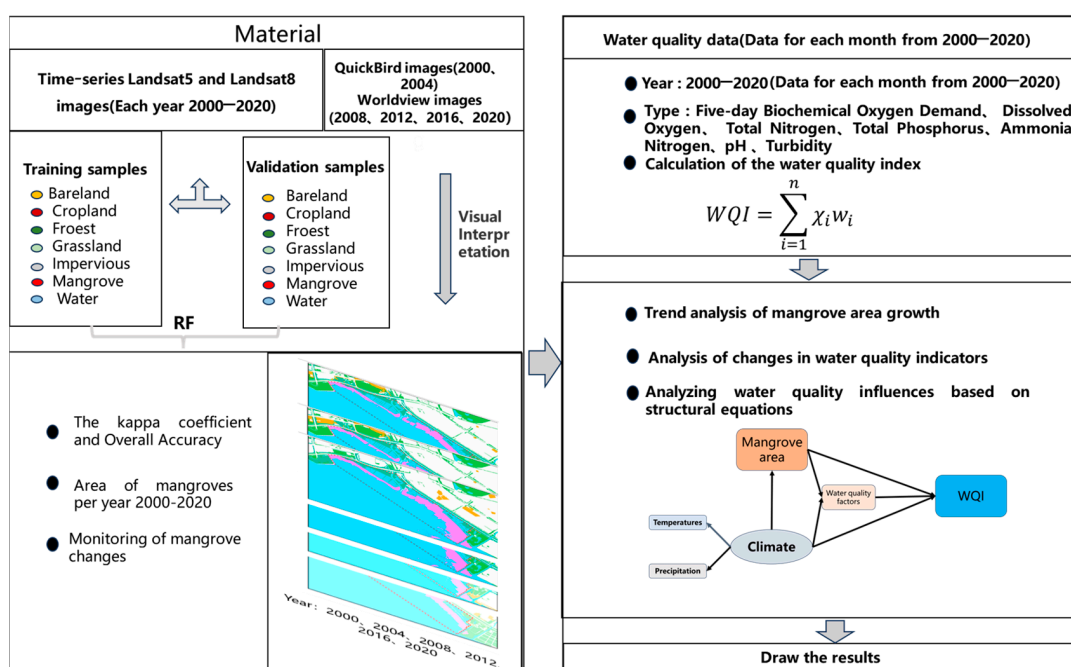

Figure 1. The workflow employed in this study.

2.2. Study Area

The FMNNR is situated on the northeast coast of Shenzhen Bay, within Futian District, Shenzhen. It spans a total area of 367.64 km², making it the smallest national nature reserve in the country, located within the city's hinterland. The reserve is influenced by water bodies originating from the Pearl River, the Shenzhen River, and the Shenzhen Bay (Reference Figure 2). The FMNNR lies in the East Asian monsoon zone, characterized by a South Asian tropical monsoon oceanic climate. The average annual temperature stands at 22.4 °C, accompanied by an average annual rainfall ranging from 1700 to 1927 mm, concentrated mainly between April and September. Evaporation averages between 1500 and 1800 mm annually, and the relative humidity is consistently around 80%. The reserve experiences approximately 2000 h of sunshine annually [38,39].
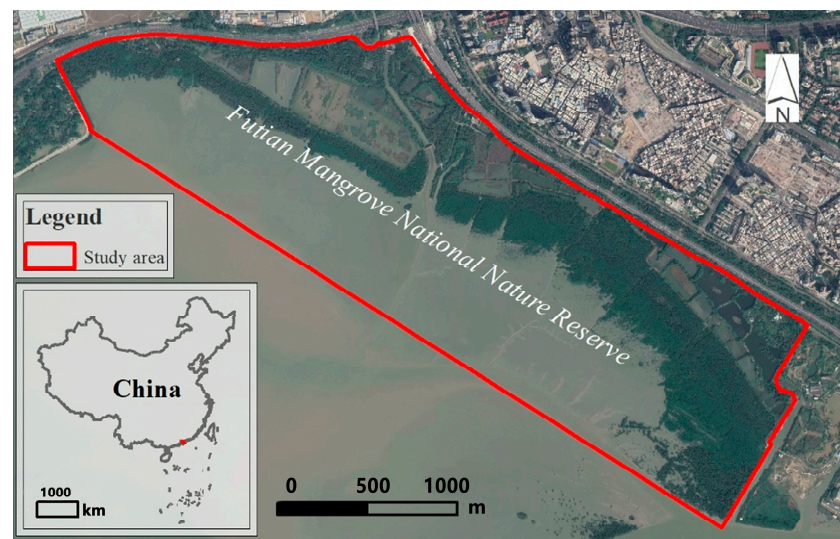

Figure 2. The location of the study area.

At the mouth of Shenzhen Bay, where the Lingdingyang tidal current enters, the bay's water depth diminishes. Simultaneously, due to topographical constraints, the tidal current exhibits a predominantly southwest-to-northeast reciprocating flow. During the rising tide, the flow direction shifts gradually toward the northeast, averaging between 258° and 40°. Conversely, the ebbs generally follow a southwesterly direction, with an average flow direction ranging from 154° to 180°. The tidal dynamics during rising and falling tides align with the waterway's direction. The tides in this region are irregular semidiurnal, characterized by significant inequality between high and low tides, with a relatively modest tidal range [40].

The mangrove forests within the Futian Reserve are primarily arranged in a belt shape, featuring uncomplicated, verdant communities comprising shrub or small tree forests. The canopy is generally well-organized, reaching an average height of 4 m, with the highest section extending up to 6 m. The study area encompasses 16 species of mangrove plants from 9 families, including 7 locally occurring mangrove species such as *Kandelia candel*, *Avicennia marina*, *Aegiceras Gaertn*, *Excoecaria agallocha* L., *Acanthus ilicifolius* L., *Bruguiera gymnorrhiza* (L.) Savigny, and others [38].

Furthermore, the mangrove reserve serves as a crucial habitat for birds and functions as a pivotal "transit point", "resting place", and "refueling station" along the north–south migration corridor for international migratory birds in the eastern hemisphere.

2.3. Dataset and Data Processing

Images from the QuickBird satellite from the years 2000 and 2004, as well as images from the WorldView satellites from the years 2008, 2012, 2016, and 2020, were employed in this study (Table 1). Given the limited availability of complete time series covering specific

locations, a decision was made to utilize post-summer images to mitigate potential variations in mangrove forests. The highest available spatial resolution for each respective image was used, corresponding to a pixel size of approximately 30 by 30 cm [37]. We geo-aligned these images due to different ranges of positional offsets. Finally, we categorized the images into seven land-use types, including mangrove, watershed, impervious cover, grassland, bare ground, and other forests. Meanwhile, to obtain more detailed area changes, we used Google Earth Engine to obtain a complementary mangrove area extraction from 2000 to 2020 after supervised classification using Landsat 5/Landsat 8 remote sensing images.

Table 1. The information of images used in this study.

ID	Year	Image Date	Bands
1	2000	1 November 2000	Red, Green, Blue
2	2004	3 October 2004	Red, Green, Blue
3	2008	20 February 2008	Red, Green, Blue
4	2012	6 November 2012	Red, Green, Blue
5	2016	19 September 2016	Red, Green, Blue
6	2020	24 November 2020	Red, Green, Blue

2.4. Potential Drive Factors Dataset

In this study, we collected water quality data from the FMNNR over the past 20 years to gain a deeper understanding of the water quality trends in the waters of Shenzhen Bay. The water quality data included seven water quality indicators, including Ammonia Nitrogen (AN), Five-day Biochemical Oxygen Demand (BOD_5), Dissolved Oxygen (DO), pH, Total Nitrogen (TN), Total Phosphorus (TP), and Turbidity (TUR) (Table 2). Besides that, to further explore the changing status of water quality we introduced a Water Quality Index (WQI). Finally, we introduced climatic data including average monthly precipitation and average monthly temperature from 2000 to 2020 to explore and interpolate monthly data from mangrove area fitting equations combined with structural equations to jointly explore the mechanisms of Futian mangrove forest and climate on water quality in Shenzhen Bay. The water quality data of Shenzhen Bay were obtained from the Environmental Protection Department of Hong Kong Special Administrative Region of China (, and these data included monthly water quality data for all years from 2000 to 2020. The monthly scale temperature data in the climate data (spatial resolution: 1 km) were obtained from the National Tibetan Plateau Scientific Data Center, while the monthly scale precipitation data were obtained from the ERA5-Land dataset (spatial resolution: $0.1° \times 0.1°$), published by the European Union and European Centre for Medium-Range Weather Forecasts (ECMWF) and other organizations.

Table 2. Climate and water quality data details.

Type	Factors	Time Available	Source
Water quality	AN BOD_5 DO pH TN TP TUR	Measurements for one day of the month, 2000–2020	Environmental Protection Department, Hong Kong Special Administrative Region, China
Climate	Temperatures Precipitation	Monthly averages, 2000–2020	China Tibetan Plateau Science Data Center; ERA5-Land dataset

Notes: Ammonia Nitrogen (AN), Five-day Biochemical Oxygen Demand (BOD_5), Dissolved Oxygen (DO), pH, Total Nitrogen (TN), Total Phosphorus (TP), Turbidity (TUR).

2.5. Image Interpretation Methods

In the domain of mangrove remote sensing, two commonly employed methods for classification are visual interpretation and supervised classification [41]. Visual interpretation allows researchers to differentiate mangrove habitats by leveraging unique geometric features and growth environments, using human visual perception to discern image characteristics such as tone and appearance [42]. On the other hand, supervised classification utilizes the random forest algorithm as an ensemble classifier based on decision trees, wherein the final estimation is made by aggregating the most frequently predicted class by individual trees [43]. The random forest algorithm has exhibited exceptional performance in land cover classification scenarios [44], and it functions as an intuitive classifier that can handle diverse types of variables and their corresponding value ranges without requiring standardization. Despite its black-box nature, random forest can assess the importance of variables, thus facilitating interpretation [45]. Accordingly, this study adopts Landsat 5 and Landsat 8 imagery acquired from the Google Earth Engine platform for the period of 2000 to 2020, employing the random forest approach for mangrove remote sensing mapping to derive the mangrove area within the Futian Mangrove National Nature Reserve (FMNNR) in Shenzhen City for each year.

2.6. Construction and Calculation of the WQI

The WQI model stands out as an excellent tool for evaluating water quality [46]. The conventional WQI model encompasses biochemical and physicochemical parameters, along with nutrient and heavy metal parameters [47–49]. Notably, studies reveal that the surface accumulation index of heavy metal elements in the coastal surface sediments of Shenzhen Bay indicates a mildly polluted and uncontaminated state [50], prompting its exclusion from the scope of this study. In the long-term mean statistical analysis of water quality factors, it was found that the concentrations of DO and BOD_5 near Shenzhen Bay are consistently at low levels, and the pH value fluctuates within a small range of around 7.5. These parameters are generally in line with the national seawater quality standards. Numerous studies underscore the significance of AN, BOD_5, DO, pH, TP, TN, and TUR as pivotal parameters in WQI computation [51–53].

Therefore, considering the specific environmental characteristics of mangrove ecosystems and in alignment with the Seawater Quality Standard, Surface Water Environmental Quality Standard, and the standards of the Bureau of Indian Standards (Table 3), we selected seven water quality parameters—AN, BOD_5, DO, pH, TP, TN, and TUR. The details of their weight determination steps are as follows: First, DO is normalized as a positive indicator, while the other indicators are normalized as negative indicators. Next, the proportion of each parameter sample, the entropy value of the parameter, and the redundancy of the information entropy of the parameter are calculated, and finally the weight of each water quality parameter is determined to establish a suitable WQI for mangroves (for more detailed steps, please refer to the Supporting Information). Formula for WQI [54]:

$$WQI = \frac{\sum_{i=1}^{n} c_i p_i}{\sum_{i=1}^{n} P_i},$$

where n is the total number of environmental parameters included in the study, C_i is the normalized value of parameter i, and P_i is the weight value of parameter i. The *WQI* value ranges from 0 to 100, and the larger the score, the better the water quality. The water quality condition was graded as Good (81~100), Fair (51~80), Marginal (31~50), and Poor (0~30) based on the *WQI* values.

Table 3. National seawater quality standards, 2002.

Water Quality Level	DO (mg/L)	BOD$_5$ (mg/L)	AN (mg/L)	TP (mg/L)	TN (mg/L)	pH	TUR (NTU)
1	7.50	3.00	0.15	0.02	0.20	7.40	1.00
2	6.00	3.00	0.50	0.10	0.50	7.80	2.00
3	3.00	4.00	1.00	0.20	1.00	8.20	3.00
4	3.00	6.00	1.50	0.30	1.50	8.60	4.00
5	2.00	10.00	2.00	0.40	2.00	9.60	5.00

Environmental parameters such as AN, DO, pH, TP, TN, and other nutrients play a critical role in shaping the composition, distribution, and growth of mangroves [55,56]. Mangroves exhibit a remarkable ability to absorb dissolved organic nutrients [57], functioning as a natural "filtration system" for efficient removal and interception of nitrogen and phosphorus. Simultaneously, DO influences microbial activity and nutrient availability in mangrove sediments. Maintaining an optimal pH range in mangrove sediments is essential for preserving their robust carbon sequestration capacity.

2.7. Construction of Structural Equations

Structural Equation Modeling (SEM) is a powerful method used to develop and test hypotheses related to relationships within a system. It encompasses a range of multivariate statistical techniques, including factor analysis, regression, path analysis, and simultaneous equation modeling [58]. SEM allows for the simultaneous investigation of causal relationships among multiple variables in complex systems, capturing indirect effects that arise from interactions between variables. Latent variables, which are constructs derived from observed variables, represent phenomena that cannot be directly observed but can be measured by several indicators (e.g., temperature, rainfall) and are therefore considered latent variables. This approach is particularly useful for studying phenomena such as climate change.

SEM is proficient in recognizing the indirect impact of mangroves on WQI changes when causal relationships are indirectly linked. For instance, climate and mangroves can influence alterations in water quality through their effects on factors contributing to water quality. The strength of SEM lies in its capacity to unravel complex relationships within a system and identify the interplay of various factors. Therefore, SEM emerges as a highly suitable tool for elucidating the influences of drivers on changes in water quality.

The procedural outline is as follows: initially, a conceptual model is constructed based on the causal or correlative relationships between variables, established upon the current comprehension of the interplay between mangroves, climate, and water quality. The focal point of our investigation revolves around assessing the influence of mangroves and climate on the fluctuation in water quality within Shenzhen Bay from 2000 to 2020. Drawing upon an extensive review of the literature concerning the driving mechanisms behind mangroves and climate's impact on water quality, our principal hypotheses are delineated (Figure 3).

Commencing from the premise of Shenzhen Bay's water quality, we postulate that sequential changes in climate and mangroves will precipitate alterations in water quality. Concurrently, climate change will exert an influence on mangrove transformations. The foundational shifts in climate and mangroves potentially exert a direct influence on changes in water quality. Furthermore, both climate change and mangrove alterations will shape the trajectory of changes in distinct water quality factors. For instance, mangrove forests possess the capacity to decelerate water eutrophication by assimilating nutrients like ammonia, total nitrogen, and total phosphorus, while also intercepting and filtering suspended substances, thereby preserving water clarity. This implies an indirect effect of underlying mangrove forests on shifts in water quality. These overarching hypotheses were transformed into a graphical conceptual model, depicting the interplay between the influencing factors and the changes in water quality within Shenzhen Bay.

The holistic fitting of the conceptual model was carried out using the entirety of the data. Any inadequate fits prompted the addition of significant missing pathways to the model, guided by directed separation principles. This iterative process continued until arriving at the conclusive model.
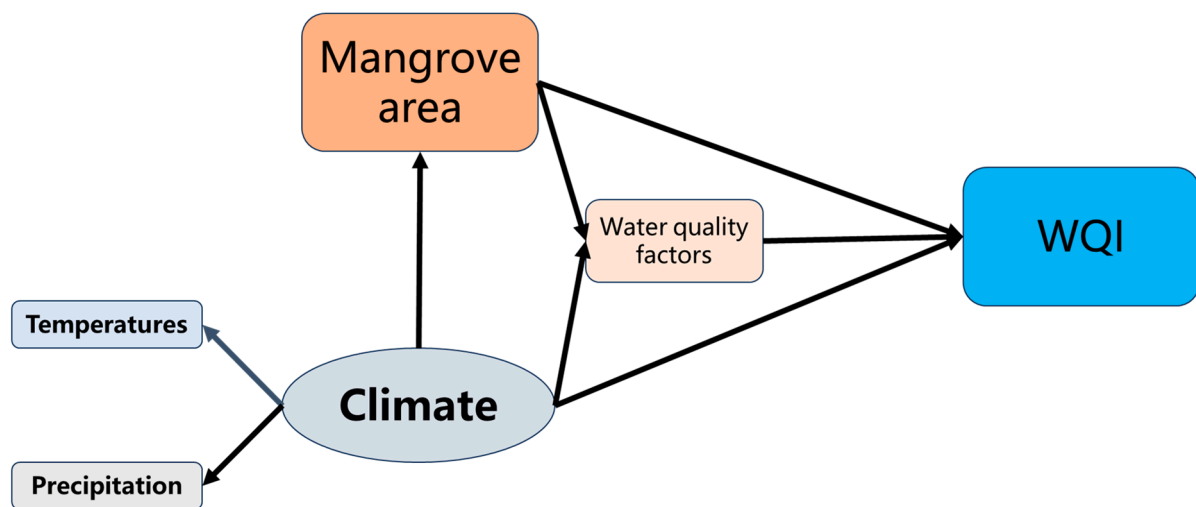
Figure 3. Conceptual model of climate and mangrove impacts on water quality.

The outcomes of Structural Equation Modeling (SEM) are conventionally portrayed in a graphical format. Here, causal associations between variables are denoted by unidirectional arrows. The arrow's origin indicates the independent variable, while its tip points towards the response variable. Additionally, bidirectional arrows signify correlations. Accompanying these arrows are standardized path coefficients, indicating the magnitude of the direct relationship between two variables, ranging from −1 to 1. A coefficient closer to 1 or −1 signifies a robust relationship, while 0 implies no relationship [59,60].

For a quantitative examination of the influence of mangroves and climate change on water quality, an SEM model was developed using 21 years of data in R.

3. Results

3.1. Land-Use Maps in FMNNR

Based on the preprocessing of images, we generated land-use maps for the FMNN for each four years from 2000 to 2020 through manual visual interpretation (Figure 4). The spatial distribution and area changes of mangrove forests during these periods are illustrated in Figure 5. Table 4 presents the area and accuracy of mangrove forests extracted through supervised classification.

The results indicate a noteworthy increase in the mangrove forest area within the FMNNR, rising from 63.907 hectares in 2000 to 106.848 hectares in 2020, reflecting a growth rate of 67.192%. The most substantial expansion occurred in the period 2016–2020, witnessing a growth of 11.268 hectares and a rate of 16.002%. Additionally, mangrove area increments for the time intervals 2000–2004, 2004–2008, 2008–2012, and 2012–2016 were 6.509 hectares, 9.277 hectares, 8.493 hectares, and 7.394 hectares, respectively, with growth rates of 10.185%, 14.457%, 11.930%, and 10.739%.

Table 4. Land-use map accuracy.

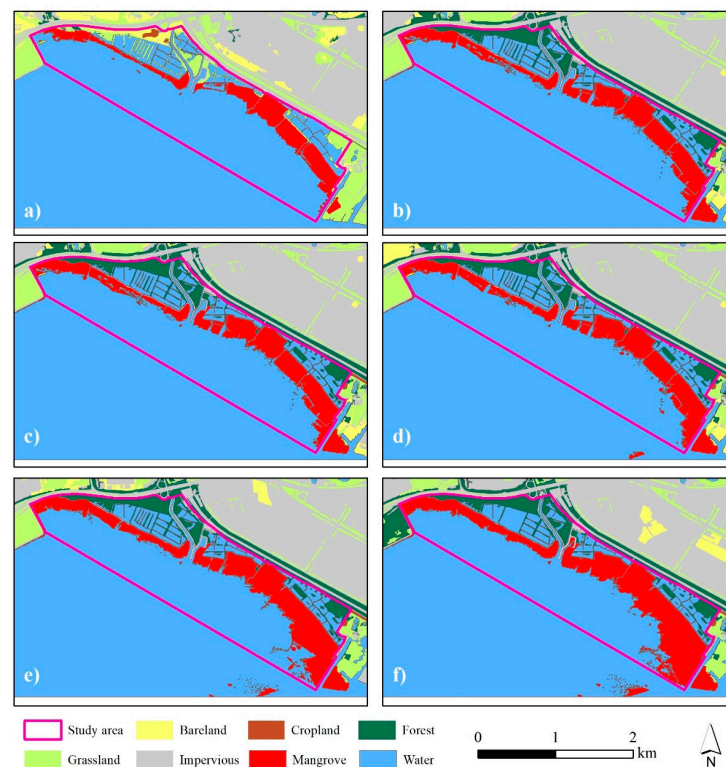| Year	Area (ha)	Kappa	Overall Accuracy
2000	63.907	0.907	0.990
2004	70.416	0.866	0.915
2008	79.693	0.822	0.893
2012	88.186	0.842	0.895
2016	95.580	0.778	0.853
2020	106.848	0.939	0.961

Figure 4. Map of the FMNNR over the past 20 years, where (**a**–**f**), each refer to the map of 2000, 2004, 2008, 2012, 2016, and 2020, respectively.
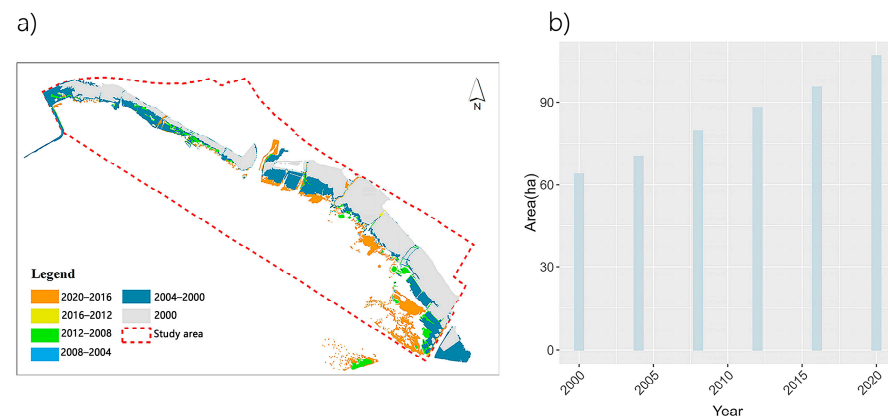

Figure 5. The spatial distribution and area change of mangrove plants in the FMNNR, where (**a**) is the spatial dynamic change of mangroves in the FMNNR and (**b**) is the histogram of mangrove forest area in the FMNNR from 2000 to 2020.

The data suggests effective protection and improvement of the mangrove environment over the past two decades within and around the reserve. Mangrove plants are extending seaward, indicating successful preservation and enhancement of their growth environment through the establishment of the Futian Mangrove National Nature Reserve.

3.2. Changes in Water Quality in Shenzhen Bay

Based on the collection of water quality data in Deep Bay, also known as Shenzhen Bay, this study examines the annual and seasonal variations as well as the correlation among water quality factors over the past two decades. Figure 6 illustrates the mean inter-annual variation in water quality in Shenzhen Bay from 2000 to 2020, while Figure 7 presents the box-mean plot of water quality during the same period. Additionally, Figure 7 offers a

box-mean plot specifically depicting water quality during periods of high water, flat water, and low water. The correlation of water quality factors in different seasons is displayed in Figure 8.
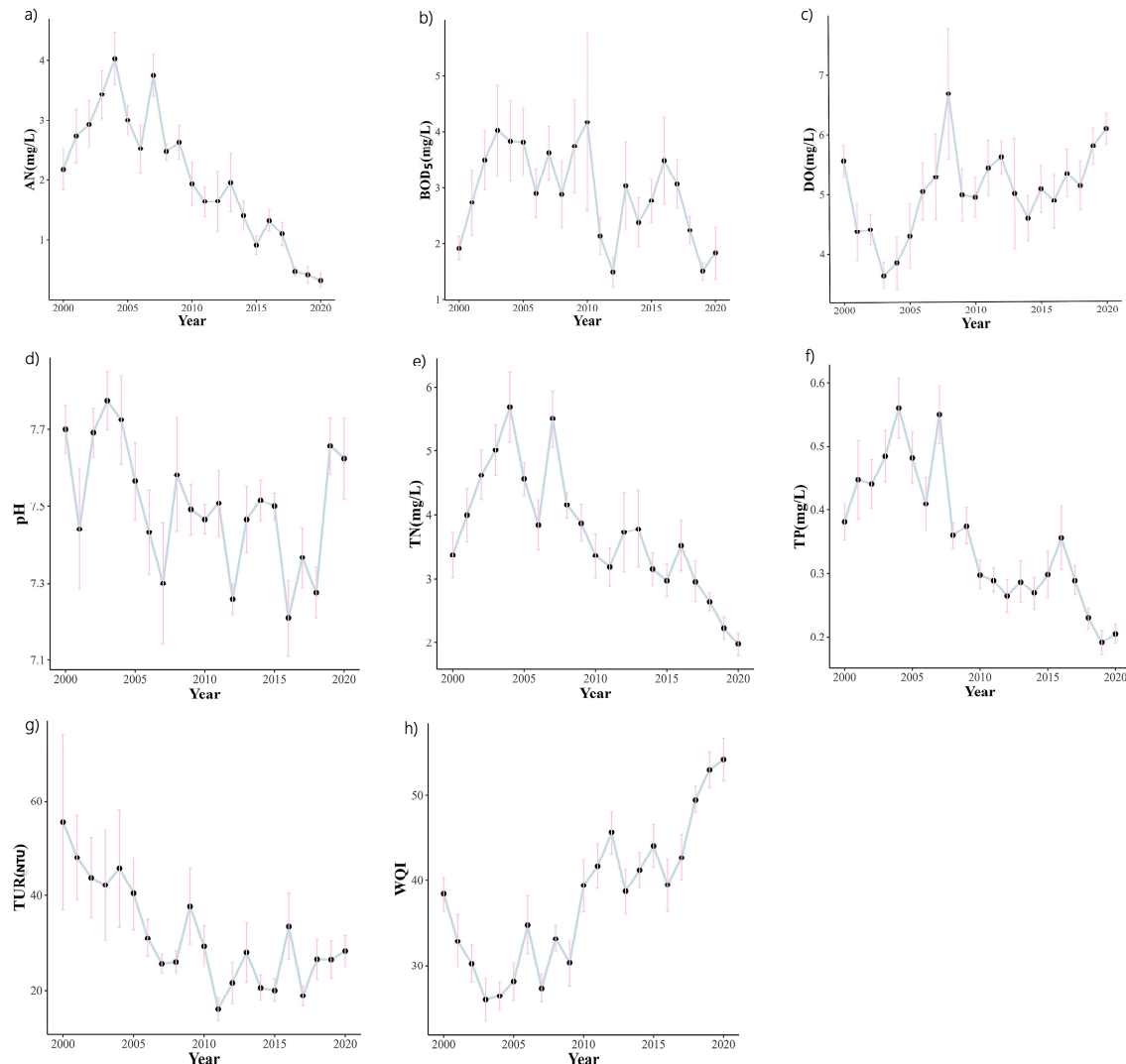

Figure 6. Line graphs of changes in major water quality factors in Shenzhen Bay from 2000 to 2020 (**a**–**g**); (**h**) is the integrated change in water quality in Shenzhen Bay from 2000 to 2020.

The comprehensive analysis of annual changes in water quality indicators in Shenzhen Bay reveals that the annual average concentrations of ammonia nitrogen, BOD_5, total nitrogen, total phosphorus, and turbidity exhibited an initial increase followed by a subsequent decrease from 2000 to 2020. In contrast, dissolved oxygen and pH demonstrated a decreasing trend followed by an increasing trend. In summary, over the period from 2000 to 2020, the water quality in Shenzhen Bay exhibited a gradual deterioration followed by an improvement.

The seasonal variations in water quality in Deep Bay exhibit distinct patterns, categorized into three periods: flat water (May and November), high water (June to October), and low water (January to April, December). These classifications were established based on interannual variations in water quantity, segmented into five time periods spanning from 2000 to 2020 for comprehensive analysis (refer to Table 5 and Figure 7 for specifics). Ammonia Nitrogen (AN) concentrations ranged from 0.30 ± 0.25 to 3.76 ± 1.14 mg/L. Between 2000 and 2008, the mean AN concentrations across each period failed to meet the standard outlined in the 2002 National Seawater Quality Standard, specifically seawater category

5 (see Table 3). The period from 2009 to 2012 marked a transitional phase, witnessing a gradual adherence to the national standard. Notably, during the high-water period within this timeframe, the mean AN concentration reached 2.33 ± 1.0 mg/L. Subsequently, from 2013 to 2020, the average AN concentration stood at 2.33 ± 1.17 mg/L across all periods, meeting the Class V seawater standard specified in the National Seawater Quality Standard of 2002.
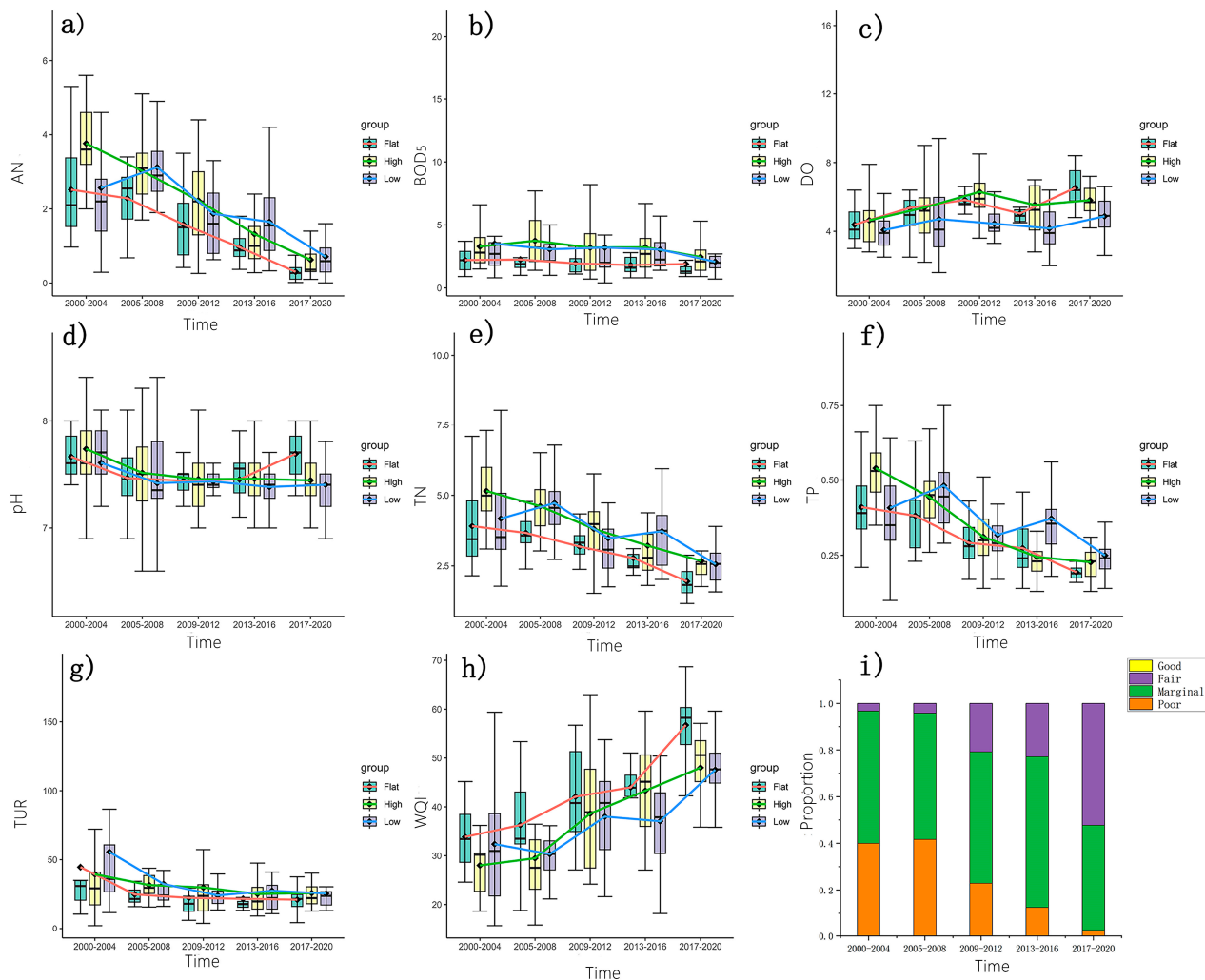

Figure 7. Box-line mean folding chart of main water quality factors in Shenzhen Bay from 2000 to 2020 (**a**–**h**); (**i**) shows the percentage of water quality classes for WQI.

Significant seasonal disparities in AN concentration were observed, with notably lower levels during flat-water periods than high-water periods (p = 0.0036). This variance in AN concentrations within the water column can be attributed to reduced precipitation and riverine inputs during low-water periods, leading to decreased water volume and heightened AN concentration. Conversely, increased precipitation during high-water periods tends to wash ammonia nitrogen from the soil into the water column, elevating AN concentrations. In contrast, flat-water periods exhibit relative stability, characterized by lower precipitation inputs into water bodies and consequently lower ammonia nitrogen levels.

Seasonal fluctuations influenced the BOD_5 concentration within the range of 1.81–3.74 mg/L, consistently meeting the standards set by seawater category IV in the 2002 National Seawater Quality Standard. Notably, BOD_5 concentrations were significantly lower during the flat-water period than during both the high-water and low-water periods (p = 0.0017, p = 0.000029). This discrepancy may be attributed to the relative stability of

the water body during flat-water periods, characterized by reduced mobility, thriving ecosystems, and enhanced efficiency in decomposing organic matter. Seasonal variations contributed to the variability in organic matter sources and distinctive water quality characteristics. Conversely, the water body experienced turbulence during high-water and low-water periods, leading to increased concentrations of ammonia and nitrogen. This turbulence limited the degradation of organic matter, resulting in higher BOD_5 concentrations.
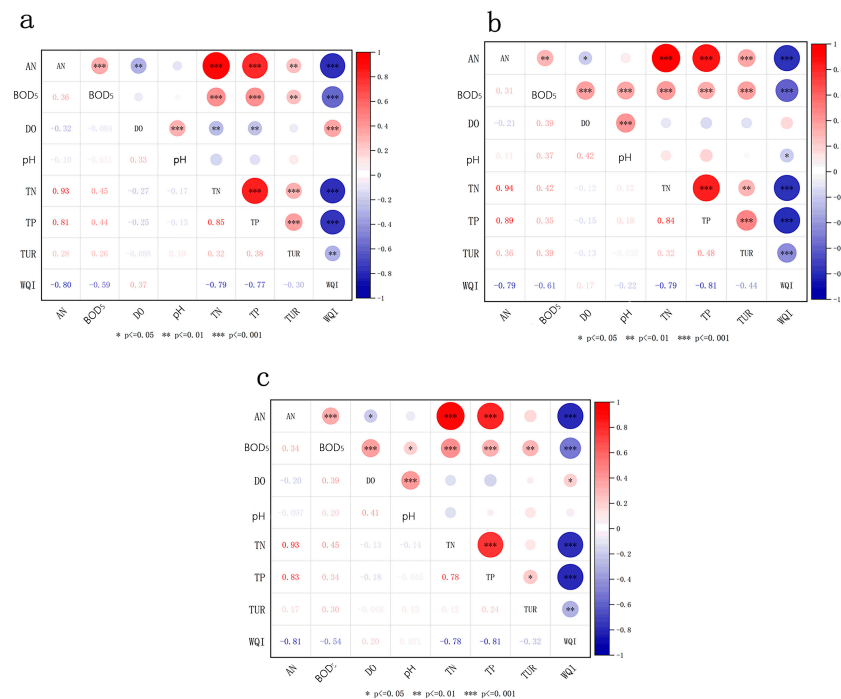

Figure 8. (**a**) High-water period; (**b**) Low-water period; (**c**) Flat-water period; Shenzhen Bay water quality change correlation.

Table 5. Water quality factor values by season, 2000–2020.

Time	Season	AN ± SD (mg/L)	BOD$_5$ ± SD (mg/L)	DO ± SD (mg/L)	pH ± SD	TN ± SD (mg/L)	TP ± SD (mg/L)	TUR ± SD (NTU)	WQI ± SD
	Flat	2.52 ± 1.39	2.21 ± 0.91	4.39 ± 1.15	7.66 ± 0.23	3.91 ± 1.53	0.41 ± 0.13	44.59 ± 49.77	33.85 ± 6.35
2000–2004	High	3.76 ± 1.14	3.28 ± 1.81	4.64 ± 1.37	7.73 ± 0.41	5.15 ± 1.13	0.54 ± 0.12	39.40 ± 35.65	28.02 ± 6.18
	Low	2.63 ± 1.62	3.60 ± 2.76	4.00 ± 1.40	7.60 ± 0.36	4.25 ± 1.91	0.41 ± 0.18	50.05 ± 45.24	31.61 ± 11.35
	Flat	2.29 ± 0.96	2.25 ± 1.39	5.36 ± 2.18	7.46 ± 0.39	3.67 ± 0.74	0.38 ± 0.13	24.79 ± 8.91	36.28 ± 10.52
2005–2008	High	3.02 ± 1.10	3.74 ± 2.02	5.38 ± 2.13	7.51 ± 0.46	4.61 ± 1.33	0.44 ± 0.12	31.58 ± 10.62	29.48 ± 9.05
	Low	3.12 ± 1.21	3.07 ± 1.28	4.71 ± 2.17	7.42 ± 0.49	4.73 ± 1.36	0.48 ± 0.17	32.34 ± 22.58	30.34 ± 6.09
	Flat	1.58 ± 1.04	1.95 ± 0.77	5.83 ± 0.63	7.44 ± 0.17	3.19 ± 0.78	0.29 ± 0.09	22.29 ± 17.31	42.08 ± 10.83
2009–2012	High	2.33 ± 1.17	3.39 ± 2.91	6.31 ± 2.85	7.47 ± 0.32	3.87 ± 1.19	0.32 ± 0.10	31.22 ± 26.55	37.52 ± 11.63
	Low	1.92 ± 1.43	3.19 ± 4.34	4.62 ± 0.93	7.41 ± 0.19	3.52 ± 1.81	0.31 ± 0.10	22.86 ± 9.81	38.39 ± 10.48
	Flat	0.97 ± 0.43	1.81 ± 0.74	5.04 ± 1.13	7.45 ± 0.32	2.78 ± 0.67	0.27 ± 0.10	21.59 ± 12.87	44.00 ± 5.53
2013–2016	High	1.31 ± 1.28	3.32 ± 2.39	5.57 ± 2.61	7.45 ± 0.26	3.22 ± 1.64	0.24 ± 0.08	25.94 ± 18.85	43.52 ± 9.59
	Low	1.64 ± 0.99	2.89 ± 2.23	4.22 ± 1.26	7.41 ± 0.25	3.68 ± 1.32	0.37 ± 0.15	26.95 ± 19.23	37.52 ± 8.23
	Flat	0.30 ± 0.25	1.90 ± 1.63	6.53 ± 1.27	7.69 ± 0.25	1.95 ± 0.58	0.19 ± 0.05	20.96 ± 10.29	56.73 ± 7.98
2017–2020	High	0.63 ± 0.68	2.46 ± 1.32	5.8 ± 0.86	7.44 ± 0.27	2.68 ± 0.88	0.23 ± 0.09	25.75 ± 11.93	48.01 ± 9.00
	Low	0.71 ± 0.53	2.08 ± 0.89	4.87 ± 1.15	7.40 ± 0.31	2.56 ± 0.71	0.25 ± 0.08	25.45 ± 13.07	47.58 ± 7.10

Throughout all seasons, the Dissolved Oxygen (DO) concentration consistently exceeded 4.00 ± 1.40 mg/L, meeting the standards for Type III seawater established by the National Seawater Quality Standard. However, during the low-water period, the DO concentration consistently lagged behind that observed in the flat-water and high-water

periods ($p = 0.00076$, $p = 0.000038$). This discrepancy can be attributed to the slowing down of water flow and reduced oxygen exchange, coupled with factors such as scarce rainfall and increased accumulation of organic wastes. The relative stability of the water body and unfavorable redox conditions for oxygen dissolution contributed to a decline in the water body's oxygen content during the low-water period, negatively impacting the aquatic ecosystem and oxygen-dependent organisms, thus maintaining a low level of DO concentration.

The pH values exhibited consistent fluctuations within the range of 7.45 to 7.73, devoid of a discernible seasonal pattern. Similarly, Total Phosphorus (TP) concentrations (ranging from 0.19 to 0.54 mg/L) and turbidity values (ranging from 20.96 to 50.05 NTU) did not display a clear seasonal trend. Although turbidity did not align with the criteria for Class V seawater stipulated in the 2002 National Seawater Quality Standards from 2000 to 2020, a noteworthy downward trend has been observed in the high-water and low-water periods ($p = 0.0129$, $p = 0.00038$). The TP concentration exceeded Class V seawater standards by only 0.5 mg/L during the high-water period from 2000 to 2004. Consequently, turbidity remains a critical indicator to be managed in the Back Bay.

TN concentrations and WQI values exhibited notable seasonal fluctuations. The TN concentration consistently registered lower values during the flat-water period compared to the high-water and low-water periods ($p = 0.0048$, $p = 0.01$). This observation can be attributed to factors such as increased turbulence within the water body, heightened nitrogen input from various sources, and restricted nitrogen cycling during the high-water and low-water phases. While the TN concentration ranged between 1.95 and 5.15 mg/L, it fell a mere 2 mg/L below the Class V seawater standard during the flat-water phase from 2017 to 2020, underscoring ongoing nitrogen pollution in the Back Bay. Notably, the TN concentration remained below the Class V seawater standard throughout the flat-water period.

The comprehensive WQI analysis revealed a consistent elevation in values during the flat-water period compared to both the high-water and low-water phases ($p = 0.013$, $p = 0.0096$), signaling a progressive enhancement in water quality within Shenzhen Bay. As depicted in Figure 7i, the monthly distribution of WQI ranks for various periods demonstrated a noteworthy shift. Specifically, between 2000 and 2004, the WQI ranks were 0.4 for poor quality and 0.033 for fair quality. In contrast, the 2017–2020 data indicated a decline in the poor-quality rank to 0.023 and an uptick in the fair-quality rank to 0.523. This evolving trend suggests a diminishing prevalence of poor water quality and an increasing trend towards fair water quality, underscoring the ongoing improvement in the water quality of Shenzhen Bay.

Through a comprehensive analysis of water quality seasonality in Shenzhen Bay spanning from 2000 to 2020, the relationship between water quality factors and the Water Quality Index (WQI) in different seasons was elucidated using Pearson's correlation coefficient (refer to Figure 8). The findings revealed that AN, BOD_5, TN, TP, and TUR exerted significant ($p < 0.01$) influences on the WQI across the flat-water, high-water, and low-water periods. Notably, AN, TN, and TP exhibited more pronounced effects on water quality in various seasons. Specifically, the correlation coefficient between AN and WQI during the flat-water period reached a substantial -0.81, while the correlation coefficient of TN in the high-water and low-water periods remained consistently high at -0.79.

Conversely, pH and DO exhibited notable variations across different seasons. The correlation between DO and WQI in the high-water period proved to be highly significant ($p < 0.001$), with a correlation coefficient of 0.37. This significance was also observed in the flat-water period, albeit to a lesser extent ($p < 0.05$), with a correlation coefficient of 0.20. Intriguingly, the correlation between DO and WQI was found to be non-significant in the low-water period. In contrast, the correlation between pH and WQI attained significance ($p < 0.05$) solely during the high-water period, with a correlation coefficient of 0.22.

Seasonal fluctuations exerted a pronounced influence on water quality, particularly during phases of elevated water levels, wherein increased water turbulence facilitated

heightened oxygen exchange, leading to elevated Dissolved Oxygen (DO) concentrations and a positive impact on overall water quality. In comparison, although the correlation was marginally weaker, it remained statistically significant during the flat-water period, signifying that seasonal variations continued to influence DO concentration during this phase. Conversely, during the low-water period, DO exhibited no significant correlation with the Water Quality Index (WQI), likely attributable to the reduced water flow impeding oxygen exchange and resulting in a decline in DO levels.

Furthermore, the noteworthy correlation between pH and WQI during low-water periods could be attributed to the relative stability of the water column and the accumulation of acidic and alkaline substances. In summary, the findings underscore the substantial impact of seasonal climate variations on water quality, particularly evident in dynamic water body conditions, such as those observed during periods of high water.

3.3. Structural Equation Analysis of the Factors Affecting Water Quality in Shenzhen Bay

The ultimate Structural Equation Model (SEM) is depicted in Figure 9, revealing robust support for interactions between the variables. The Goodness of Fit (GOF) indices for the final SEM are presented in Table 6. Notably, all fitted structural equations exhibit a Chisq/Df ratio of less than 2, *p*-values exceeding 0.5, and perfect Comparative Fit Indices (CFIs) of 1, while the Root Mean Square Error of Approximation (RMSEA) values are uniformly 0.000. These results collectively signify a well-fitted model, attesting to the robustness of the specified relationships.
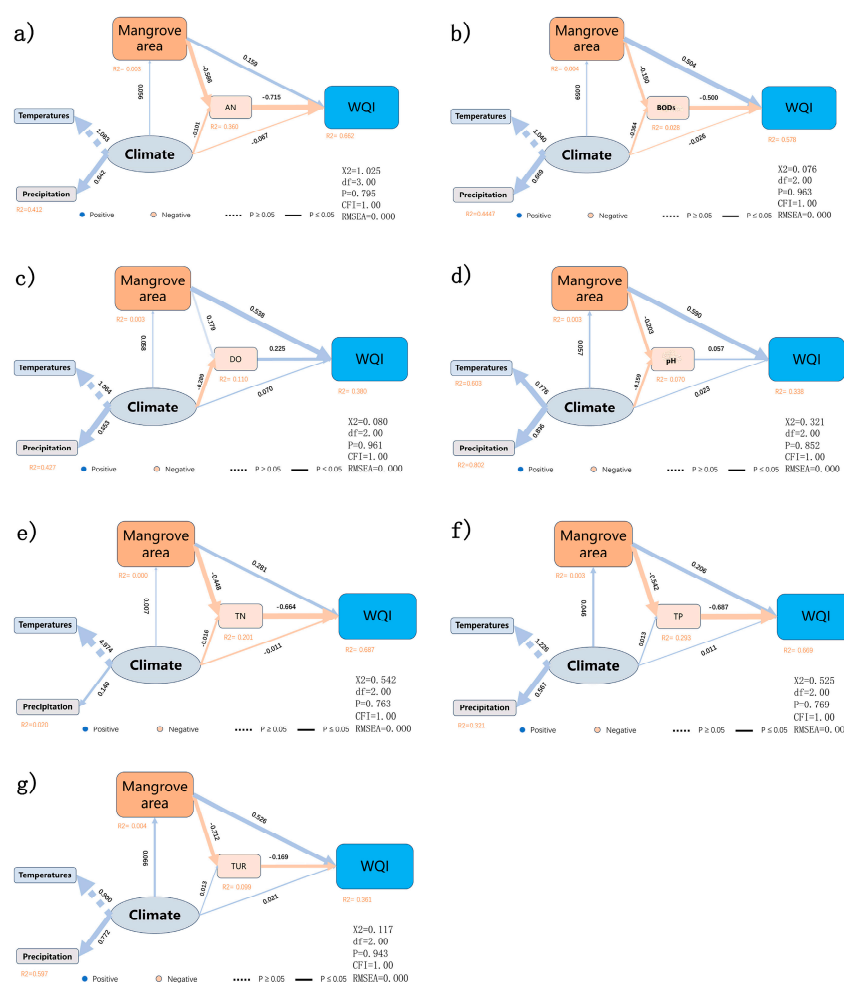

Figure 9. Multivariate relationship between changes in WQI and its drivers: (**a**), (**b**), (**c**), (**d**), (**e**), (**f**), and (**g**) respectively illustrate how climate and mangrove area influence water quality through their impacts on AN, BOD_5, DO, pH, TND, TP, and TUR.

Table 6. Structural equation fitting metrics.

Water Quality Factors	Chisq	Df	*p*-Value	CFI	RMSEA
AN	1.025	3.000	0.795	1.000	0.000
BOD$_5$	0.076	2.000	0.963	1.000	0.000
DO	0.080	2.000	0.961	1.000	0.000
pH	0.321	2.000	0.852	1.000	0.000
TN	0.542	2.000	0.763	1.000	0.000
TP	0.525	2.000	0.769	1.000	0.000
TUR	0.117	2.000	0.943	1.000	0.000

The findings reveal that alterations in the Water Quality Index (WQI) are influenced by climate change, changes in mangrove areas, and variations in different water quality factors to varying extents (Figure 8). Ammonium Nitrogen (AN), Biochemical Oxygen Demand (BOD$_5$), Total Nitrogen (TN), Total Phosphorus (TP), and Turbidity (TUR) exerted a negative and direct impact on WQI changes in Shenzhen Bay, with impact coefficients of -0.715, -0.5, -0.664, -0.687, and -0.169, respectively. Conversely, Dissolved Oxygen (DO) and pH had positive and direct impacts on WQI changes in Shenzhen Bay, with impact coefficients of 0.225 and 0.057, respectively. The positive direct impact of WQI changes in Shenzhen Bay is evident, with impact coefficients of 0.225 and 0.057 for DO and pH, respectively. Therefore, AN emerges as the key negative impact factor for WQI changes in Shenzhen Bay, followed by BOD$_5$, TN, TP, TUR, and DO, all of which play significant roles in shaping WQI in Shenzhen Bay. pH has a relatively minor impact on WQI changes in Shenzhen Bay, possibly attributed to the relatively small fluctuations in pH within the water body of Shenzhen Bay and the inherent buffering capacity of the water body.

Climate change and the mangrove area exhibit not only a direct impact on WQI changes in Shenzhen Bay but also exert an indirect influence (Figures 9 and 10). Upon comparing the direct and indirect effects of mangrove forests on WQI in Shenzhen Bay, it is evident that the indirect effects surpass the direct effects. When the mangrove forest area traverses indirect paths involving AN, TN, and TP, leading to changes in WQI in Shenzhen Bay, the associated indirect effect coefficients are noteworthy, measuring 0.419, 0.297, and 0.372, respectively.

This observation implies that mangrove forests contribute significantly to water quality by absorbing ammonia nitrogen, total nitrogen, total phosphorus, and other nutrients. They also play a crucial role in intercepting and filtering suspended substances, maintaining water transparency, and impeding the progression of water eutrophication. The root system of mangroves serves to stabilize the soil and prevent sediment input, thereby contributing to the reduction of water body turbidity. The positive impact on biodiversity and the regulation of ecological balance in the ecosystem further enhance the water quality of Shenzhen Bay. These intricate mechanisms enable changes in the mangrove area to influence the Shenzhen Bay Water Quality Index (WQI), primarily through its effects on parameters such as AN, TN, and TP, thereby sustaining the health of coastal waters.

On the flip side, there exist three indirect pathways from climate change to the alteration of the Water Quality Index (WQI) in Shenzhen Bay. Two of these indirect paths originate from variations in the mangrove area and involve subsequent changes in WQI. However, owing to the adaptability of mangrove forests, their slow growth, and the root structure's resistance to rising sea levels, the impact of climate on mangrove area changes is minimal. The coefficients representing the direct influence of climate-induced changes on the mangrove area are less than 0.07. Consequently, the indirect effects of climate on the two paths above leading to WQI changes in Shenzhen Bay are below 0.05. The indirect impacts of climate on these paths to the WQI of Shenzhen Bay are thus low, with coefficients of indirect influence on WQI changes being less than 0.04.
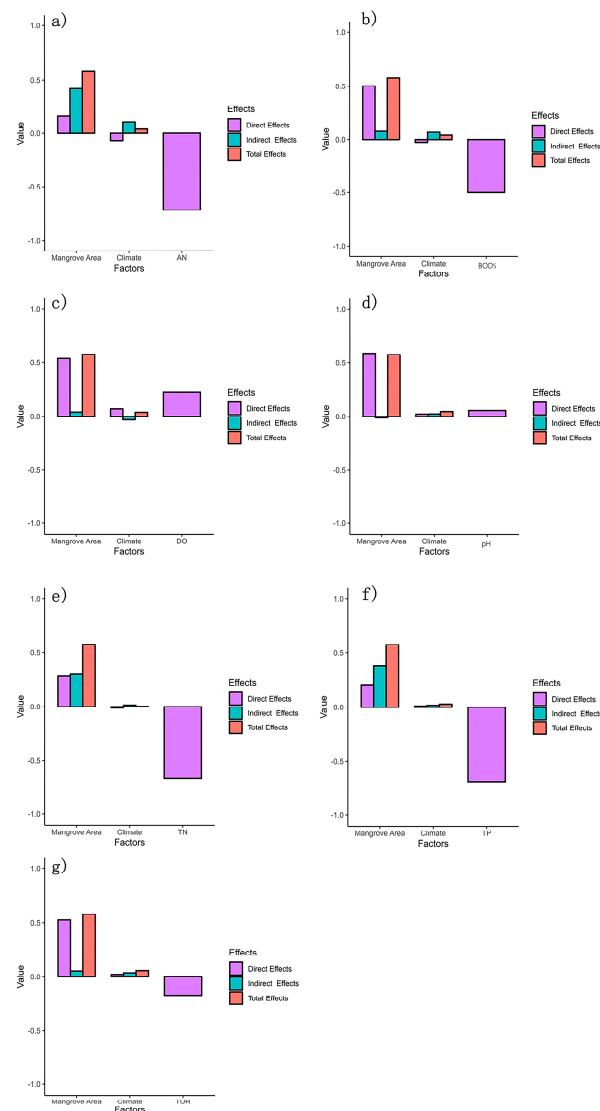
Figure 10. Direct, indirect, and overall effects of climate and mangrove area on water quality through different drivers such as AN (**a**), BOD5 (**b**), DO (**c**), pH (**d**), TN (**e**), TP (**f**) and TUR (**g**).

The third indirect path operates through alterations in each water quality factor influencing the overall change in WQI in Shenzhen Bay. This stands out as the primary indirect route through which climate influences the water quality of Shenzhen Bay. This influence is particularly pronounced in Ammonia Nitrogen (AN) and Dissolved Oxygen (DO), with their respective indirect influence coefficients on WQI changes in Shenzhen Bay being 0.072 and -0.065.

In the comprehensive assessment of the direct, indirect, and overall impacts of climate, mangrove area, and water quality factors on WQI changes in the SEM model, our findings reveal that Climate exerts a relatively lesser impact compared to mangrove areas on the water quality of Shenzhen Bay. Examining the direct impact, the minimum coefficient representing the direct influence of mangrove area change on WQI change is 0.159, significantly surpassing the maximum coefficient of direct impact attributed to climate-induced changes in the WQI, which is 0.07. Furthermore, in the context of the indirect impact of mangrove area change on WQI through various water quality factors, the coefficients of indirect impact via AN, BOD$_5$, DO, TN, TP, and TUR to the WQI of Shenzhen Bay are greater than the corresponding indirect influence factor of climate on them. In conclusion, the overall influence of mangrove area changes on WQI in Shenzhen Bay far exceeds the cumulative influence of climate change on it.

4. Discussion

4.1. The Positive Impact of Mangrove Expansion on Water Quality

From 2000 to 2020, the mangrove area of the FMNNR exhibited a consistent expansion trend, surging from 63.907 hectares in 2000 to 106.848 hectares in 2020, reflecting a remarkable growth rate of 67.192%. The most substantial increase occurred during the 2016–2020 period, witnessing an expansion of 11.268 hectares and a growth rate of 16.002%. Moreover, the mangrove area witnessed increments in the four distinct time spans periods of 2000–2004, 2004–2008, 2008–2012, and 2012–2016, with respective expansions of 6.509 hectares, 9.277 hectares, 8.493 hectares, and 7.394 hectares, accompanied by growth rates of 10.185%, 14.457%, 11.930%, and 10.739%.

Over the past two decades, mangrove plants within and around the reserve have received enhanced protection, evidenced by their expansion seaward. This outcome suggests that delineating the boundaries of the Futian Mangrove National Nature Reserve has mitigated human interference, leading to the effective preservation and enhancement of the mangrove plant environment. For example, since 1998, the continuous reforestation project in Shenzhen has provided a solid foundation for the expansion of mangroves [61]. It is undeniable that changes in the area of mangroves may also be related to variations in factors such as industrial emissions, road construction, and differences in mangrove species. The enlarging mangrove range holds the potential for manifold benefits to water quality [62–64]. Firstly, the mangrove trees and root systems boast exceptional ecological filtration capacity, efficiently extracting deleterious substances from water bodies, including deposited sludge and waste [65–67]. Secondly, mangrove forests act as natural protective barriers, thwarting the ingress of terrestrial pollutants into water bodies and sustaining pristine water quality [68–70]. Additionally, mangroves serve as optimal habitats for numerous aquatic organisms, fostering diversity in watershed ecosystems and upholding ecological equilibrium [71,72]. In summary, the expansion of mangrove areas offers vital ecological services for water quality, contributing significantly to the establishment of a robust and unpolluted water environment.

4.2. Factors Contributing to Water Quality Improvement in Shenzhen Bay

From 2000 to 2004, the Water Quality Index (WQI) registered values of 0.4 for poor and 0.033 for fair. In contrast, from 2017 to 2020, there was a reduction in the percentage of poor to 0.023, coupled with an increase in the percentage of fair to 0.523. The constructed WQI exhibited its low value at 26.013 in 2003 and its peak at 54.164 in 2020. Analyzing the annual change trend (refer to Figure 6), Shenzhen Bay's water quality underwent a phase of degradation followed by amelioration.

Between 2000 and 2003, concentrations of ammonia, BOD_5, total phosphorus, and total nitrogen in the water displayed an upward trajectory. However, from 2004 to 2020, the concentrations of these water quality factors have gradually decreased. The WQI values for Shenzhen Bay manifested a general pattern of decrease followed by an increase over time. The initial deterioration in water quality from 2000 to 2004 may be attributed to the combined impacts of urbanization, industrialization, and meteorological conditions [73]. The subsequent improvement in water quality can be linked to the successive implementation of environmental protection policies by the government, the gradual expansion of the mangrove area, and an enhancement in societal awareness regarding environmental protection [74,75].

A plethora of studies indicate that the primary source of water pollution in Shenzhen Bay is domestic sewage from upstream areas [76–79]. However, numerous other pollutants still indirectly affect the water quality of Shenzhen Bay. For instance, large-scale construction projects before 2010 played a role in deteriorating water quality, and with the completion of these projects the transport of pollutants associated with demolition and construction activities to Shenzhen Bay has decreased, thereby benefiting the improvement of water quality. Additionally, although there is no external pollution input into the fishponds within the FMNNR, local fishermen previously cultivated fish in these ponds before

2006 [80]. During this period, a significant amount of organic matter as well as nitrogen and phosphorus compounds accumulated in the sediment. When the dissolved oxygen concentration in the water decreases (typically in January), nitrogen and phosphorus are released back into the water from the sediment, leading to an increase in nitrogen and phosphorus levels within the fishponds [81,82]. Lastly, in April and May 2011, Shenzhen City initiated standardized inspections of pollution treatment facilities. Enterprises lacking pollution control facilities meeting regulatory standards were mandated to rectify their practices within a specified timeframe, significantly curtailing the discharge of industrial wastewater [83]. However, with the continual increase in the discharge of untreated sewage and considering Shenzhen Bay's semi-enclosed nature and poor water exchange dynamics, with water residence times ranging from 15 to 25 days [84], pollutants discharged into the bay have difficulty exiting, resulting in the accumulation of pollutants.

Additionally, water quality factors such as ammonia, BOD_5, DO, and TN exhibit distinct seasonal patterns (refer to Table 5 and Figure 6). Specifically, during the same period, ammonia nitrogen, BOD_5, and TN consistently maintained low levels during the flat-water period, while DO consistently remained low during the low-water period. This observation may be attributed to the relative stabilization of the water body in the flat-water period, leading to reduced biological activities. Conversely, the low levels of DO during the low-water period may result from a combination of factors, including increased biological oxygen demand and reduced water flow.

To further elucidate the impact of different water quality factors on water quality across seasons, correlation analysis was employed. The results revealed significant variations in pH and DO across different seasons (see Figure 7). Notably, there was a highly significant correlation between DO and WQI during the high-water period ($p < 0.001$), with a correlation coefficient of 0.37. Additionally, a more pronounced correlation was observed during the flat-water period ($p < 0.05$), with a correlation coefficient of 0.20. However, the correlation between DO and WQI during the low-water period (refer to Figure 8) was not significant. Furthermore, the correlation between pH and WQI attained significance ($p < 0.05$) solely in the high-water period, with a correlation coefficient of 0.22 [85,86].

These findings suggest that different seasons exert diverse effects on water quality, with climate emerging as a pivotal factor during flat, dry, and high-water periods. Climate directly influences the water quantity of the water body by impacting precipitation, evaporation, and seasonal variations, thus establishing its significance in shaping the overall quality of the water body.

4.3. Comparison of Impacts on Water Quality between Mangroves and Climate

In certain studies, examining the impact of mangroves on water quality, limited attention has been given to the intricate mechanisms underlying changes in water quality. Structural Equation Modeling (SEM) proves to be a valuable method for investigating the causal network between factors, encompassing alterations in mangrove areas, shifts in water quality factors, and the influence of climate. In our investigation, we conceptualized underlying climate change as a latent variable. Through the incorporation of latent variables, we quantitatively assessed the relationship between changes in the mangrove area and alterations in each water quality factor [87–89].

Furthermore, SEM facilitated the elucidation of both the direct and indirect effects of influencing factors on changes in the Water Quality Index (WQI) within Shenzhen Bay. Our findings revealed that both mangrove area and climate exerted influence on individual water quality factors, while each water quality factor, in turn, impacted changes in WQI. Consequently, mangrove area and climate exhibited indirect effects on WQI changes. This not only aligns with our comprehension of water quality change mechanisms but also bears a resemblance to results corroborated by some prior studies.

The influence of each water quality factor on changes in WQI varies, leading to different impacts of climate change and mangrove areas on water quality alterations through distinct factors. As depicted in Figure 9 of the SEM model, mangrove areas and

climate change demonstrate diverse responses to various water quality factors, thereby exerting differing indirect effects on WQI. Notably, AN, BOD$_5$, TP, and TN emerge as pivotal factors influencing water quality.

Mangrove forests exhibit a robust indirect effect on WQI by strongly responding to AN, TN, and TP, resulting in indirect effects of 0.419, 0.297, and 0.372, respectively. In contrast, the indirect effect of climate change is relatively modest, primarily manifesting through its impact on WQI via AN, DO, and pH responses.

Irrespective of direct, indirect, or cumulative effects, changes in mangrove areas wield a significantly greater influence on water quality compared to climate change, largely due to their potent and direct ecological filtration. In contrast, climate change impacts water temperature and ammonia nitrogen levels, albeit through relatively indirect pathways. Consequently, the direct and comprehensive ecological contributions of alterations in mangrove areas significantly enhance water body health.

4.4. Implications

Our study presents a robust framework for a more comprehensive understanding and elucidation of the intricate interrelationships underlying long-term changes in water quality within the framework of mangrove expansion and climate change. We established a Water Quality Index (WQI) to characterize the specific state of water quality, employing Structural Equation Models (SEMs) to delve into the intrinsic mechanisms driving changes in water quality within the context of both mangrove areas and climate change. This framework facilitates the prediction of water quality alterations, elucidates the intricate interactions among water quality, mangroves, and climate, and proposes targeted measures for environmental protection and management of water quality.

Our findings contribute valuable insights for scientific research and the formulation of policies about water quality and mangrove forests. Moreover, they play a crucial role in advancing the sustainable management of water ecosystems.

4.5. Limitations and Directions for Future Research

This study has several limitations. Firstly, the sample data used for calibrating the monthly mangrove area model consists of interpolated data rather than direct measurements. Secondly, the relationship between influencing factors and changes in water quality is typically nonlinear. However, quantifying such intricate nonlinear relationships is challenging, and our study utilized linear estimates in the Structural Equation Model (SEM). Future research should explore potential nonlinear relationships between influencing factors and changes in water quality. Thirdly, the fitted model did not fully capture the variability observed in water quality changes. This discrepancy may be attributed to unmeasured factors, necessitating further investigation. Nonetheless, the constructed model exhibited good explanatory power, as indicated by the R squared and fitness index.

Finally, due to the absence of controlled experiments, the results of this study may exhibit some bias. Future research can enhance rigor by incorporating control groups, thereby enabling a more accurate assessment of the impacts of various factors on water quality. Additionally, we have not yet considered factors such as wastewater discharge, construction activities, and physiological parameter variations among mangrove species, which could significantly affect water quality. Therefore, future studies should take these potential influencing factors into account more comprehensively to achieve a more holistic evaluation of water quality.

5. Conclusions

In this study, we conducted a comprehensive analysis of the long-term changes in water quality in the Futian National Nature Reserve and Shenzhen Bay from 2000 to 2020. Firstly, the mangrove forests in the Futian National Mangrove Reserve exhibited a trend of expansion towards the ocean during this period. Secondly, the seasonal and annual characteristics of water quality in Shenzhen Bay indicated a significant improvement

in water quality from 2000 to 2020. Lastly, through structural equation modeling, we unveiled the impact pathways, direct effects, and indirect effects of Shenzhen Bay's climate, mangroves, and water quality factors on water quality. The results demonstrated that mangroves have significant direct and indirect impacts on water quality, which are more pronounced than the effects of climate factors. Future research could further explore the relationship between the species, morphology, and physiological characteristics of mangroves and water quality, providing a more detailed quantification of the causal relationship between mangroves and water quality.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/f15071246/s1, Table S1: Parameters for indices calculation in the water quality index (WQI); Table S2: The normalized value of Ci.

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Data Availability Statement: Data available in a publicly accessible repository that does not issue DOIs. Publicly available datasets were analyzed in this study. This data can be found here: https://www.epd.gov.hk/epd/sc_chi/top.html, accessed on 11 June 2024.

Conflicts of Interest: The authors declare no conflicts of interest.

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