EDITOR

Prof. Dr. İsmet DAŞDEMİR

Researches and Evaluations in the Field of AQUACULTURE

December
2024

İmtiyaz Sahibi / Yaşar Hız
Yayına Hazırlayan / Gece Kitaplığı
Birinci Basım / Aralık 2024 - Ankara
ISBN / 978-625-388-111-5

Gece Kitaplığı
Kızılay Mah. Fevzi Çakmak 1. Sokak
Ümit Apt No: 22/A Çankaya/ANKARA
0312 384 80 40
www.gecekitapligi.com / gecekitapligi@gmail.com

Baskı & Cilt
Bizim Büro
Sertifika No: 42488

RESEARCHES AND EVALUATIONS
IN THE FIELD OF
AQUACULTURE

EDITOR
Prof. Dr. İsmet DAŞDEMİR

gece kitaplığı

CONTENTS

CHAPTER 1

CHAPTER 2

CHAPTER 1

CARRYING CAPACITY IN AQUACULTURE

Birol BAKİ[1], Oylum GÖKKURT BAKİ[2]

---

[1] Prof. Dr., Sinop University, Faculty of Fisheries and Aquatic Science, Department of Aquaculture, Sinop, Turkey (ORCİD Id: 0000-0002-2414-1145) bbaki@sinop.edu.tr

[2] Assoc. Prof. Dr., Sinop University Faculty of Engineering and Architecture, Environmental Engineering Department, Sinop, Turkey (ORCİD Id: 0000-0001-7823-0824) ogbaki@sinop.edu.tr

Introduction

The global population is projected to reach approximately 10 billion by 2050 and 11.2 billion by 2100 (UN-WPP, 2024). This demographic growth is expected to significantly increase the demand for animal-based food products, underscoring the pivotal role of the aquaculture sector in providing safe and high-quality food. Aquaculture production is anticipated to expand substantially in the coming years to address the rising demand for animal protein driven by population growth (Duarte et al., 2009). Cultured fish, as a reliable and nutritious protein source for human consumption, is indispensable in meeting the nutritional needs of the growing population. However, expanding the aquaculture sector has led to conflicts among coastal area users and raised environmental concerns. Competition between marine cage systems and other coastal activities, in particular, has prompted the relocation of production areas to alternative or offshore sites. While this transition offers opportunities for expanding usable surface areas, it also introduces significant challenges.

The perception of aquaculture has frequently been unfavorable due to concerns over product quality and the environmental consequences of this practice. It is crucial to identify, evaluate, and incorporate these concerns into the processes of site selection and management for aquaculture activities (Yucel-Gier, 2017).

The selection of suitable sites for marine cage systems, assessment of carrying capacity, and implementation of effective governance mechanisms are critical for achieving sustainable aquaculture. Environmental impact assessment processes must consider factors such as water quality standards, the impact of coastal waste on marine ecosystems, current velocity, and discharge rates (Ross et al., 2013a; Zhu and Dong, 2013). Within this framework, adopting an ecosystem-based approach is fundamental to ensuring the sustainability of aquaculture practices.

Comprehensive analyses of the ecological and social impacts of production areas are essential for accurately determining specific marine zones' carrying capacities (Stelzenmüller et al., 2017). Since the 1960s, ecosystem-based management approaches have emphasized ecological capacity by incorporating production potential into planning processes (Weitzman et al., 2019). This transition represents a critical step toward enhancing aquaculture's environmental sustainability and ensuring reliable food sources in the long term.

As a result, to enable the global aquaculture sector to contribute effectively to food security, it is imperative to adopt an ecosystem-based approach, develop and enforce sustainable practices, and ensure that regulatory authorities mandate such practices. Controlled production, guided by the determination of individual production areas' carrying capacities, is essential for achieving these goals.

Environmental Effects of Aquaculture

Coastal ecosystems, particularly areas suitable for aquaculture, have been subjected to increasing environmental pressures in recent years. In this context, assessing aquaculture environments' carrying capacities and adopting robust environmental regulations through best practices are indispensable prerequisites for the sustainable development of aquaculture and production.

Yuningsih et al. (2014) highlighted that fish excretions and uneaten feed are the primary sources of organic matter generated by floating net cages. Such accumulation of organic matter can deteriorate water quality, ultimately leading to the death of fish and other aquatic organisms. This degradation negatively impacts biodiversity and compromises the sustainability of aquaculture operations. Consequently, the density of floating net cages is a critical factor; a higher number of cages results in a greater accumulation of organic matter (Gorlach-Lira et al., 2013; Syandri et al., 2016).

The primary environmental issues caused by cage-based marine fish farming waste are categorized as diffuse or point-source pollution. The formation of fish farm waste and its environmental impacts depend on several factors, including fish size, water temperature, current direction and speed, farming methods, feed composition, feeding strategies, species farmed, stocking density, and the nutrient concentrations in the receiving water body (Pillay, 2004; Yıldırım and Korkut, 2004; Fisheries and Oceans Canada, 2006). Aquaculture operations have the potential to influence various environmental factors, including water quality, plankton, benthos, nekton, biomass, and species diversity. The severity of these impacts depends on factors such as farming methods, production scale, local hydrological conditions, stock density, farm management practices, feed composition, and the biological, chemical, and physical attributes of the farming area (Davies, 2000; Beveridge, 2004; Yıldırım and Korkut, 2004).

Before establishing fish farms, it is essential to determine the nutrient concentrations, dynamics of the water body, and nutrient emissions per unit area to evaluate the environmental impacts of waste from cage systems. The total waste load and distribution in the aquaculture zone are also critical (Chary, 2021). As the aquaculture sector rapidly expands, feed production and consumption have reached significant levels. Consequently, farm solid and dissolved waste output is expected to increase proportionally with production (Cho et al., 1994).

Waste generated by cage-based fish farming is associated with significant environmental concerns, such as the formation of anoxic conditions in aquatic and sediment environments, as well as algal blooms. This waste includes organic solids, dissolved waste, and inorganic nutrients. When the influx of these substances into the surrounding ecosystem surpasses the natural carrying capacity, it can result in environmental issues such as eutrophication, oxygen depletion, and alterations in biodiversity within both the water column and the seabed (IUCN, 2007).

In marine fish farming, phosphorus is the critical limiting factor for primary productivity, and its excess can lead to eutrophication. Coastal marine areas with rich endemic fauna and flora often exhibit low biomass and abundance due to oligotrophic conditions. Research has shown minimal increases in chlorophyll-a levels in the water column near production areas (Pitta et al., 1999; Nordvarg and Johansson, 2002; La Rosa et al., 2002; Soto and Norambuena, 2004; Pitta et al., 2005). Using bioassays with macroalgae and phytoplankton, Dalsgaard and Krause-Jensen (2006) demonstrated that primary productivity, which is higher near cages, decreases sharply with increasing distance from fish farms. Other studies in intensive farming areas have reported anoxia, the presence of *Beggiatoa spp.*, and the absence of macrofauna (Rosenthal and Rangeley, 1988; Hansen et al., 1991; Holmer and Kristensen, 1992; Karakassis et al., 2000; Tomassetti and Porrello, 2005; Klaoudatos et al., 2006; Yucel-Gier et al., 2007; Dimitriadis and Koutsoubas, 2008).

Regarding the EU Water Framework Directive, Aguado-Giménez et al. (2006) and Karakassis et al. (2000) argued that the benthic quality in areas hosting fish farms cannot be classified as "High" or "Good" regardless of the applied index. Various factors contributing to the decline of seagrasses (*Posidonia oceanica*) due to aquaculture impacts have been investigated, revealing that sedimentation of farm waste is the primary driver of benthic degradation (Holmer et al., 2008).

In addition to environmental impacts, marine cage farming zones have been reported to serve as protected areas, leading to increased fish abundance, biomass, species richness, and egg productivity due to factors such as the absence of fishing, the provision of high-protein and high-fat feeds, and minimized starvation risks (Dempster et al., 2002; Golani, 2003; Smith et al., 2003; Vega Fernandez et al., 2003; Machias et al., 2004; Vita et al., 2004; Machias et al., 2006; Tuya et al., 2006; Fernandez-Jover et al., 2007). Studies have attributed ecosystem changes to shifts in primary production driven by the rapid dissolution of dissolved nutrients (feed waste and excreta) in oligotrophic regions experiencing nutrient scarcity.

When assessing the environmental impacts of cage-based fish farming, it is crucial to consider the effects on sediment and water column nutrient levels and the resultant eutrophication processes. The global, regional, and local environmental effects of intensive fish farming in cages are widely recognized (Folke and Kautsky, 1989).

Carrying Capacity in Aquaculture Production

The trends in aquaculture practices in marine environments are discussed based on current and up-to-date global data. Information on farmed species, production systems, potential aquaculture areas, and production yields per unit area suggests a growing shift toward offshore aquaculture, supported by coastal integration frameworks. This shift is expected to foster the sector's continued development.

The increase in aquaculture production today is driven by the expansion of production areas, the growth of operating enterprises, and enhancements in production capacities. Large-scale enterprises predominantly carry out marine aquaculture. The shared nature of marine areas among coastal nations limits individual countries' unilateral action, necessitating collaborative resource utilization approaches.

Estimating the carrying capacity for fish aquaculture typically involves modeling the maximum permissible production, primarily by assessing potential environmental changes. Key factors include nutrient inputs or extractions and variations in oxygen levels, depending on the species being farmed. These assessments are conducted for specific catchment areas or water bodies, considering the number of aquaculture units involved. In extractive operations like shellfish farming, primary concerns are food depletion and the consequent effects on wild species and their food sources (Aguilar-Manjarrez et al., 2017).

Ecological carrying capacity refers to an ecosystem's ability to maintain its functions while integrating aquaculture activities, ensuring that environmental quality standards are upheld. This concept, sometimes termed assimilative capacity, indicates the system's capability to absorb certain levels of nutrients or oxygen consumption without adverse outcomes such as eutrophication. Aquaculture introduces dissolved and particulate matter into the environment, consumes oxygen and other resources, and may release residues from disease treatments or chemicals. Evaluating the ecological carrying capacity involves analyzing these impacts on the ecosystem. Factors such as water depth, flushing rates or current velocity, temperature, and biological activity in both the water column and bottom sediments influence the capacity of a given area. Due to the multifaceted nature of ecological capacity, models are often employed to integrate these various factors and assess their combined effects (Aguilar-Manjarrez et al., 2017).

It's also crucial to consider existing waste inputs into shared water bodies from sources like sewage discharges, agricultural runoff, domestic waste, and forestry. The cumulative impact of all aquaculture operations and background inputs should be compared with the ecosystem's ecological capacity to determine the sustainable extent of aquaculture within a specific area. However, diffuse inputs, as opposed to point sources, are challenging to assess and measure, complicating the estimation of their current effects. Additionally, long-standing activities such as agriculture or forestry may have already influenced current water quality and conditions, reflecting their historical impacts.

Aquaculture has emerged as a sector with the potential to contribute significantly to sustainable food production worldwide. However, its sustainability is contingent on several factors, with aquatic ecosystems' carrying or carrying capacity being among the most critical. Carrying capacity in aquaculture refers to evaluating production limits, environmental constraints, and social acceptability (Ross et al., 2013b). It also emphasizes the nutrient input levels aquatic ecosystems can handle without posing a risk of eutrophication (Ganguly et al., 2015). The concept encompasses various interrelated definitions, including physical, social, and ecological carrying capacities, each addressing distinct management goals and interactions between aquaculture and surrounding ecosystems (Weitzman and Filgueira, 2020).

Exceeding the ecological carrying capacity in aquaculture can lead to several environmental issues, such as eutrophication, increased primary productivity, and nutrient-induced phytoplankton blooms. These phenomena often result from nutrient runoff from aquaculture operations. Additionally, the accumulation of harmful sediments, including fish feces and uneaten feed, can degrade habitat quality and reduce biodiversity. For aquaculture farmers, these environmental challenges can cause significant losses, including fish stock mortality due to algal blooms, oxygen depletion, and disease outbreaks. Notably, fish cage culture systems, which are open to the surrounding environment, extract oxygen from the water and release waste materials directly into adjacent waters and sediments, thereby influencing local ecosystems (Aguilar-Manjarrez et al., 2017).

Carrying capacity denotes the maximum number and biomass of organisms an ecosystem can sustain within a specific timeframe. Accurate calculation of carrying capacity in aquaculture ensures economic efficiency and environmental balance. While overfishing can disrupt ecosystem equilibrium, improper aquaculture practices may exceed the carrying capacity of water resources, leading to adverse impacts on natural balance, species extinction, and water quality degradation, ultimately threatening human health. Thus, carrying capacity indicates not only the ecological resilience of a system but also the extent to which natural resources can be sustainably exploited.

Addressing the overuse of natural resources requires research and practical applications. Understanding the potential of fish species, wetlands, and water resources increases this capacity. Assessing the carrying capacity of aquaculture involves analyzing factors such as the nutritional needs of aquatic organisms, water quality, ecosystem balance, and production methods. Sustainable aquaculture aims to maintain optimal carrying capacity to preserve biodiversity and minimize environmental impacts. Strategies include implementing advanced cultivation techniques, utilizing transformative feed sources, and adopting ecosystem management practices. Additionally, local, national, and international policies must support resource-efficient practices and enhance carrying capacity. Adhering to these principles is critical for achieving economic gains while ensuring environmental sustainability. Steps taken in this regard will shape the future of a healthy and sustainable aquaculture system.

Considering carrying capacity is vital for balancing aquaculture's economic efficiency and ecological health. This approach preserves current resources and ensures a healthy environment for future generations. Increased scientific research and education are essential for the advancement of aquaculture. Determining carrying capacity requires diverse scientific methods, considering water temperature, oxygen levels, nutrient availability, and species interactions. Aquaculture enterprises must integrate these considerations into their planning processes.

The planning and managing of aquaculture areas are critical for the future of environmentally friendly aquaculture businesses (Borg et al., 2011). Management protocols and decision-makers must prioritize appropriate site selection, environmental impact monitoring, and adherence to carrying capacity criteria (Macias et al., 2019; Weitzman and Filgueira, 2020; Weitzman et al., 2021; Yiğit et al., 2021). This approach promotes sustainable growth in cage aquaculture while harmonizing marine industry development and ecosystem management (Yiğit et al., 2024).

Estimating production capacity in aquaculture regions requires extensive field measurements, analyses, and environmental monitoring. In situations demanding rapid managerial decisions, practical methods for quickly and reliably assessing carrying capacity are necessary to avoid adverse effects on marine ecosystems. These efforts are crucial for establishing long-term environmental monitoring procedures to ensure sustainable development in aquaculture enterprises (Yiğit et al., 2021).

The concept of carrying capacity, well-defined in ecology, is described as the maximum population size of a species that an environment can sustain over an extended period. The growing demand among stakeholders, including managers and producers, for estimating carrying capacity reflects positive advancements in aquaculture.

In cage aquaculture, water use may seem unlimited in practice. However, issues arise in determining carrying capacity when oxygen levels are insufficient, or water renewal rates are low. In cases of excessive organic enrichment, anoxia problems may occur, necessitating a temporary halt in aquaculture activities or a reduction in production volumes before reaching critical depths. Hence, determining carrying capacity based on environmental criteria-setting production levels that do not cause ecological degradation-is essential. Environmental impacts vary based on the characteristics of the production area (e.g., the physicochemical properties of seawater), variables defining the system's carrying capacity, and management practices, including feed efficiency (feed conversion ratios), fecal waste, and other production-related impacts (e.g., solid waste in the area).

In the early stages of aquaculture development, mass-balance modeling was employed to assess the ecological carrying capacity of freshwater lakes. A notable example is the adaptation by Dillon and Rigler (1974) of Vollenweider's (1968) phosphorus-based model, which posited that phosphorus concentrations could predict phytoplankton growth and subsequent eutrophication. This approach involved evaluating phosphorus inputs from fish farming activities to predict potential impacts on water quality. The model has been extensively utilized to determine the carrying capacity of lakes for aquaculture, including applications in Chile. Subsequent adaptations have considered nitrogen as the limiting nutrient, as discussed by Soto et al. (2007). Karakassis et al. (2013) utilized a rapid assessment model for cage aquaculture's physical and ecological carrying capacity. This model aims to determine the maximum tolerable production levels based on marine area conditions and various production factors (Macias et al., 2019; Yiğit et al., 2021).

McKindsey et al. (2006) delineate four categories of carrying capacity-physical, production, ecological, and social-that can be prioritized differently depending on the specific region and aquaculture system. These categories align with the three principal objectives of the Ecosystem Approach to Aquaculture (EAA): human well-being, ecological well-being, and effective governance. The social carrying capacity, in particular, addresses the socio-economic and governance goals of the EAA. The relative significance of each carrying capacity category may vary across different regions or cultural systems and is subject to change over time based on societal feedback. Nonetheless, harmonizing the three EAA objectives remains essential for the enduring sustainability of aquaculture (Ross et al., 2013a).

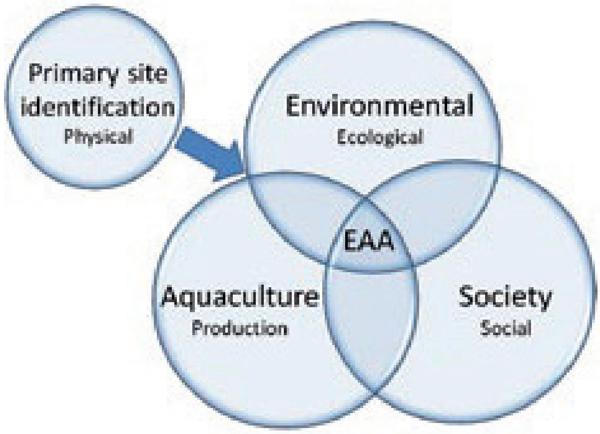

Figure 1. *Interaction of different site identification categories and carrying capacity factors in achieving an ecosystem-based approach to aquaculture (Ross et al., 2013a).*

Ecological carrying capacity models synthesize hydrodynamic, biogeochemical, and ecological processes within the environment, encompassing oxygen dynamics and the fluxes of organic matter and nutrients resulting from aquaculture activities, to assess their impact on ecosystem health. Numerous studies have been conducted to determine aquaculture's environmental impacts and monitor the process over time. Some of these include EcoWin, ACExR-LESV (Loch Ecosystem State Vector), Modelling-Ongrowing fish farms-Monitoring (MOM), MERAMED, MedVeg, AQCESS, BIOSSS, SAMI EU FP6, ECASA EU FP6, SPICOSA AB FP6, and PREVENT-ESCAPE.

EcoWin: Combines hydrodynamic models with changes to water biogeochemistry to look at large-scale, multiyear changes under non-aquaculture and aquaculture conditions (Ferreira, 1995),

ACExR-LESV: Resolve seasonal variations in oxygen and chlorophyll in defined sea areas (Tett et al., 2011),

The Modelling–Ongrowing fish farms–Monitoring (MOM): This system is designed to assess the environmental impact of fish farms at both local and regional levels. It includes a module that evaluates water quality and oxygen concentration on a broader scale (Stigebrandt, 2011),

MERAMED: A study aimed at developing monitoring guidelines and modeling tools for aquaculture's environmental impacts (benthic effects) (Black et al., 2001; Karakassis et al., 2013),

MedVeg: Focused on the impacts of nutrient release from aquaculture on benthic vegetation in coastal ecosystems (Karakassis et al., 2013),

AQCESS: Addressed both environmental and socio-economic aspects of aquaculture, including coastal use conflicts, workforce potential, and regional large-scale analyses (Karakassis et al., 2013),

BIOSSS: Assessed the use of floating biofilters and substrate effects to mitigate the impacts of aquaculture,

SAMI EU FP6: Investigated the ecological risks associated with fishmeal and fish oil production processes for aquaculture feed, alongside the effects of aquaculture on marine ecosystems,

ECASA EU FP6: Developed indicators shared by multiple experts for creating environmental models within the ecosystem approach to sustainable aquaculture,

SPICOSA AB FP6: Conducted studies integrating science and policy for coastal system evaluations, including the aquaculture sector,

PREVENT-ESCAPE: Examined strategies to reduce fish escapes from cages, their environmental impacts, and pollution associated with aquaculture.

These studies are designed to identify the effects of aquaculture, establish thresholds for ecological change before reaching unacceptable levels, and apply these thresholds. Groffman et al. (2006) describe an ecological threshold as the juncture at which minor alterations in an environmental variable lead to substantial changes within an ecosystem. Similarly, Hassan (2006) characterizes legal thresholds as the point where pollution levels become unacceptable, thereby distinguishing between acceptable and unacceptable pollution. In this framework, defining environmental quality standards and thresholds is essential for assessing a production site's carrying capacity and facilitates effective environmental impact evaluations and monitoring activities.

Carrying Capacity Estimation

A critical concept for aquaculture's development and sustainability is a system's carrying capacity (McKindsey et al., 2006). It is widely acknowledged that all human production activities, including aquaculture, have limits beyond which adverse effects occur. Carrying capacity refers to determining the quantity or density of fish farming in cages that do not lead to adverse environmental impacts across a broader area.

Carrying capacity plays a critical role in ecosystem functioning and human activities. As a natural solvent, water facilitates the dissolution and transport of various substances through rivers, lakes, and oceans, which is essential for aquatic ecosystem balance and human activities like agriculture and industry. Water flow rate, temperature, density, and the type and amount of dissolved substances influence carrying capacity. However, human activities can negatively impact this capacity. Industrial waste, agricultural chemical pollution, and other contaminants reduce water quality, limiting carrying capacity. This degradation can disrupt aquatic habitats and pose risks to human health. Thus, sustainable water resource management is crucial for maintaining ecosystem balance and human well-being. Ensuring and enhancing carrying capacity is vital today and a significant responsibility for future generations.

As aquaculture production scales up, particularly in the near future, the sector will require updated roadmaps for site selection and carrying capacity estimation. Potential aquaculture areas should aim to:

1. Increase production,
2. Minimize conflicts with other sectors, and
3. Improve production models to reduce environmental impacts.

The Working Group on Site Selection and Carrying Capacity (WGSC) highlights that the absence of an Ecological Quality Status (EQS) framework and variability in monitoring practices expose the aquaculture sector to conflicts with coastal users and accusations of environmental degradation. Thus, WGSC emphasizes the need for protocols agreed upon by stakeholders and criteria for site selection.

Carrying capacity analysis involves holistically evaluating all human activities, environmental variables, and their thresholds. It defines the total production volume or intensity for the species being cultivated. Production scale is closely linked to site size, distance from the shore, depth, and currents, as represented in the formula below (Karakassis, 2013; Karakassis et al., 2013).

$\text{Carrying Capacity} = [150 + 80 * (E-1)] * fA * fB * fK$

E = Fish production area (hectares),

fA = Distance from shore coefficient,

fB = Depth coefficient,

$fK = M / (G1+G2) * M / (L1+L2)$

M = The narrowest opening of the gulf towards the sea,

$(G1+G2)$ = The sum of the main topographic axes of the gulf, (L1+L2) = The sum of the distances from the two points of the axis (M) that determines the width of the bay to the center of the fish production area.

The coefficients fA, fB, and fK have different values depending on the characteristics of each production area.

Examples of calculating the fK coefficient (Fig 2, 3).

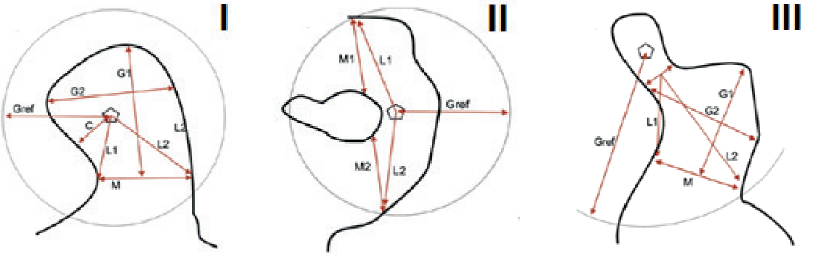

Figure 2. *Parameters used for calculating indicator F in areas with different topographical characteristics (Karakassis et al., 2013).*

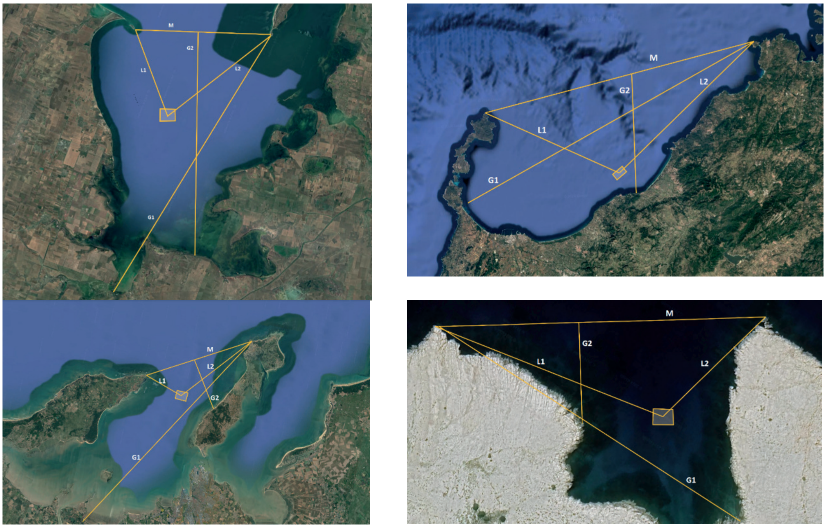

Figure 3. *Examples of the fK coefficient calculations*

Conclusion

The ecological carrying capacity for finfish aquaculture refers to the maximum level of production that can be maintained without causing significant alterations to local ecological processes, species, populations, or communities (Byron and Costa-Pierce, 2013). Environmental concerns related to finfish farming primarily stem from nutrient inputs, such as uneaten feed and fish waste, which elevate levels of phosphorus, carbon, and nitrogen in the surrounding environment.

Due to these nutrient inputs, local water quality deteriorates and leads to sediment accumulation beneath fish farms. In severe cases where the standing stock exceeds the ecological carrying capacity, environmental sustainability is compromised (Mayerle et al., 2017). These accumulations, in particular, lead to significant accumulations on the sediment surface.

Determining the carrying capacity of coastal ecosystems or open water systems presents significant challenges due to the intricate interplay of oceanographic and biological conditions and the absence of distinct boundaries. Various modeling techniques have been developed for these environments, ranging from those capable of predicting changes across extensive areas to others that assess the localized impacts of individual aquaculture operations, such as fish or mussel farms, with potential for broader application.

In this context, the "Integrated Coastal Zone Management (ICZM)" framework emphasizes preventing the ecological carrying capacity from being exceeded. ICZM principles advocate for comprehensive monitoring and analytical processes to identify natural and anthropogenic stressors in marine environments and address conflicts of interest (Kapetsky and Aguilar-Manjarrez, 2007). Coastal management strategies within this framework focus on delineating current and planned uses of coastal zones, understanding their interactions, and addressing key issues in coastal governance. Such strategies incorporate protective and precautionary measures, including the pre-assessment and ongoing monitoring of major project impacts. They also promote integrating national resources and environmental accounting methods to quantify changes in value associated with pollution, resource depletion, and habitat degradation, as well as reflect the broader environmental costs of coastal and marine area use (Gökkurt-Baki, 2017). These measures, supported by ICZM, contribute to enhancing the ecological carrying capacity of marine systems.

Geographic Information Systems (GIS) have emerged as indispensable tools for environmental analysis and management, offering capabilities in data acquisition, storage, organization, visualization, reporting, and spatial analysis and modeling (Kapetsky & Aguilar-Manjarrez, 2007). In ecological carrying capacity studies, GIS is instrumental in identifying physical constraints, such as water depth and proximity to existing activities. This includes establishing minimum distances between aquaculture sites and other sensitive areas, ensuring adequate spacing, and verifying sufficient water depth and circulation. For instance, in the United States, the EcoWin model has been integrated with other tools in Chesapeake Bay and Puget Sound to assess ecological and community capacity at the fish farm level (Bricker et al., 2013; Saurel et al., 2014). Similar modeling projects have been conducted in Portugal (Ferreira et al., 2014) and Ireland (Nunes et al., 2011).

In regions lacking sophisticated regional models, simpler approaches can be adopted to manage production within acceptable limits. For example, in the Philippines, aquaculture is restricted to 5% of the water body, though this does not directly predict carrying capacity. In Norway, aquaculture development between 1996 and 2005 was managed through feed quotas, which indirectly controlled production by limiting feed distribution. This system incentivized farmers to optimize feed use, thereby improving feed conversion ratios (FCRs) and reducing environmental impacts. Additional regulations, such as a maximum cage volume of 12,000 m^3 per license and density limits, complemented this approach. Over time, Norway transitioned to direct assessments of carrying capacity, conducted in situ or at smaller spatial scales, to refine aquaculture management practices.

Indices such as the Trophic Index (TRIX) are instrumental in assessing the ecological status of aquatic systems, particularly in evaluating the impact of aquaculture on eutrophication levels. In Turkey, the TRIX index has been applied to monitor eutrophication potential in various water bodies, providing insights into nutrient dynamics and guiding sustainable aquaculture practices.

In Scotland, predictive modeling approaches have been employed to estimate nutrient enhancement and benthic impacts resulting from fish farming activities. These models inform locational guidelines by identifying areas that are environmentally sensitive to further aquaculture development due to elevated nutrient levels or significant benthic effects. Determining the carrying capacity for aquaculture becomes increasingly complex over larger regions due to the multitude of interacting dynamic factors and the acceptable thresholds for environmental change. Adopting an ecosystem-based approach is essential, emphasizing the implementation of comprehensive management principles, utilization of appropriate tools, engagement with other sectors, and fostering stakeholder participation alongside the development of incentive structures.

References

Aguado-Giménez, F., García-García, B., Hernández-Lorente, M.D., Cerezo-Valverde, J., 2006. Gross metabolic waste output estimates using a nutritional approach in Atlantic bluefin tuna (*Thunnus thynnus*) under intensive fattening conditions in western Mediterranean Sea. Aquacult. Res., 37: 1254–1258.

Aguilar-Manjarrez, J., Soto, D., Brummett, R., 2017. Aquaculture zoning, site selection and area management under the ecosystem approach to aquaculture. A handbook. Report ACS18071, Food and Agriculture Organization of the United Nations, and The World Bank, Rome. Download part 1 pdf, full report pdf; hard copies can be purchased through publications-sales@fao.org. Aquaculture Environment Interactions, 5: 255–270.

Beveridge, M., 2004. Cage Aquaculture, third edition. Oxford, UK, Blackwell Publishing Ltd. 368 pp.

Black, K.D., Pearson, T.H., Kögeler, J., Thetmeyer, H., Karakassis, I., 2001. MERAMED: Development of monitoring guidelines and modelling tools for environmental effects from Mediterranean aquaculture. In: Uriarte A. (ed.), Basurco B. (ed.). Environmental impact assessment of Mediterranean aquaculture farms. Zaragoza: CIHEAM, 2001. p. 201-203 (Cahiers Options Méditerranéennes; n. 55).

Borg, J.A., Crosetti, D., Massa, F., 2011. Site selection and carrying capacity in Mediterranean marine aquaculture: key issues (WGSC-SHoCMed). In: General Fisheries Commission for the Mediterranean, Thirtyffth Session edn, Rome Italy, p 180 (9-14 May 2011). http://gfcmsitestorage.blob.core.windows.net/documents/web/GFCM/35/GFCM_XXXV_2011_Dma.9.pdf.

Bricker, S., Ferreira, J.G., Zhu, C., Rose, J., Galimany, E., Wikfors, G., Saurel, C., Miller, R.L., Wands, J., Wellman, K., Rhealt, R., Getchis, T., Tedesco, M., 2013. The FARM model in Long Island Sound: how important is nutrient removal through shellfish harvest? Chesapeake Bay Program Modeling–Quarterly Review Meeting, 9–10 April 2013. (also available at www.chesapeakebay.net/channel_files/18874/suzanne_bricker_-_the_farm_model_in_long_ island_sound-how_important_is_nutrient_removal_through_shellfish_harvest_041013.pdf).

Chary, K., Callier, M.D., Cove`s, D., Aubin, J., Simon, J., Fiandrino, A., 2021. Scenarios of fish waste deposition at the sub-lagoon scale: a modelling approach for aquaculture zoning and site selection. ICES Journal of Marine Science, doi:10.1093/icesjms/fsaa238.

Cho, C.Y., Hynes, J.D., Wood, K.R., Yoshida, H.K., 1994. Development of high-nutrient-dense, low-pollution diets and prediction of aquaculture wastes using biological approaches. Aquaculture, 124: 293-305.

Dalsgaard, T., Krause-Jensen, D., 2006. Monitoring nutrient release from fish farms with macroalgal and phytoplankton bioassays. Aquaculture, 256: 302-310.

Davies, P., 2000. Cage Culture of Salmonids in Lakes: Best Practice and Risk Management for Tasmania.

Dempster, T., Sanchez-Jerez, P., Bayle-Sempere, J.T., Gimenez-Casalduero, F., Valle, C., 2002. Attraction of wild fish to sea-cage fish farms in the south-western Mediterranean Sea: Spatial and short-term temporal variability. Mar. Ecol. Prog. Ser., 242: 237-252.

Dillon, P.J., Rigler, F.H., 1974. The phosphoruschlorophyll relationship in lakes. Limnology and Oceanography, 19: 767-773.

Dimitriadis, C., Koutsoubas, D., 2008. Community properties of benthic molluscs as indicators of environmental stress induced by organic enrichment. J. Nat. Hist., 42: 559-574.

Duarte, C.M., Holmer, M., Olsen, Y., Soto, D., Marbà, N., Guiu, J., Black, K., Karakassis, I., 2009. Will the oceans help feed humanity? Bioscience, 59:967-976.

Fernandez-Jover, D., Jimenez, J.A.L., Sanchez-Jerez, P., Bayle-Sempere, J., Casalduero, F.G., Lopez, F.J.M., Dempster, T., 2007. Changes in body condition and fatty acid composition of wild Mediterranean horse mackerel (*Trachurus mediterraneus*, Steindachner, 1868) associated to sea cage fish farms. Mar. Environ. Res., 63:1-18.

Ferreira, J.G., 1995. EcoWin–an object-oriented ecological model for aquatic ecosystems. Ecological Modelling, 79: 21–34. (also available at www.longline.co.uk/site/products/aquaculture/ecowin).

Ferreira, J.G., Saurel, C., Lencart e Silva, J.D., Nunes, J.P., Vasquez, F., 2014. Modelling interactions between inshore and offshore aquaculture. Aquaculture, 426-427: 154-164.

Fisheries and Oceans Canada, 2006. Canadian Technical Report of Fisheries and Aquatic Sciences 2450.

Folke, C., Kautsky, N., 1989. The role of ecoshystems for a sustainable development of aquaculture development. Ambio, 18: 234-243.

Ganguly, D., Patra, S., Muduli, P.R., Vardhan, K.V., Abhilash, R.K., Robin, R.S., Subramanian, B.R., 2015. Infuence of nutrient input on the trophic state of a tropical brackish water lagoon. J. Earth Syst. Sci. 124:1005-1017. https://doi.org/10.1007/s12040-015-0582-9.

Gillibrand, P.A., Gubbins, M.J., Greathead, C. Davies, I.M., 2002. Locational guidelines for fish farming: predicting levels of nutrient enhancement and benthic impact. Scottish Fisheries Research Report No. 63/2002. (also available at www.gov.scot/Uploads/Documents/Report63.pdf).

Golani, D., 2003. Fish assemblages associated with net pen mariculture and an adjacent rocky habitat in the Port of Ashdod Israel (eastern Mediterranean)-preliminary results. Acta Adriat., 44: 51-59.

Gorlach-Lira, K., Pacheco, C., Carvalho, L.C.T., Junior, H.N.M., Crispim, M.C., 2013. The influence of fish culture in floating net cages on microbial indicators of water quality. Brazilian Journal of Biology 73(3):457-463.

Gökkurt-Baki, 2017. The Coastal Zone Management in Turkey the Principles and Basic Problems, pp:2163-2168, Chapter: 243, Researches on Science and Art ın 21st Century Turkey, Gece Publishing, Editors: Prof. Hasan Arapgirlioğlu, Assist. Prof. Atilla Atik, Prof. Robert L. Elliott, Assoc. Prof. Edward Turgeon, Vol. 2.

Groffman, P., Baron, J., Blett, T., Gold, A., Goodman, I., Gunderson, L., Levinson, B., Palmer, M., Paerl, H., Peterson, G., Poff, N., Rejeski, D., Reynolds, J., Turner, M., Weathers, K., Wiens, J., 2006. Ecological thresholds: the key to successful environmental management or an important concept with no practical application? Ecosystems 9, 1-13.

Hansen, P.K., Pittman, K., Ervik, A., 1991. Organic waste from marine fish farms-Effects on the seabed. Marine Aquaculture and Environment., 22: 105-119.

Hassan, D., 2006. Protecting the Marine Environment from Land-based Sources of Pollution: Towards Effective International Cooperation. Ashgate Publishing Ltd, Aldershot, UK 41.

Holmer, M., Argyrou, M., Dalsgaard, T., Danovaro, R., Diaz-Almela, E., Duarte, C.M., Frederiksen, M., Grau, A., Karakassis, I., Marba, N., Mirto, S., Perez, M., Pusceddu, A., Tsapakis, M., 2008. Effects of fish farm waste on *Posidonia oceanica* meadows: Synthesis and provision of monitoring and management tools. Mar. Pollut. Bull., 56:1618-1629.

Holmer, M., Kristensen, E., 1992. Impact of fish cage farming on metabolism and sulfate reduction of underlying sediments. Marine Ecology Progress Series. 80:191-201.

IUCN (The World Conservation Union), 2007. Guide for the Sustainable Development of Mediterranean Aquaculture No:1.

Kapetsky, J.M., Aguilar-Manjarrez, J., 2007. Geographic information systems, remote sensing and mapping for the development and management of marine aquaculture. FAO Fisheries Technical Paper. No. 458. Rome. FAO. 125 pp. (www.fao.org/docrep/009/a0906e/a0906e00.HTM).

Karakassis, I., 2013. Environmental interactions and initiatives on site selection and carrying capacity estimation for fish farming in the Mediterranean. In: L.G. Ross, T.C. Telfer, L. Falconer, D. Soto, J. Aguilar-Manjarrez, eds. Site selection and carrying capacities for inland and coastal aquaculture, pp. 161-170. FAO/Institute of Aquaculture, University of Stirling, Expert Workshop, 6-8 December 2010. Stirling, the United Kingdom of Great

Britain and Northern Ireland. FAO Fisheries and Aquaculture Proceedings No. 21. Rome, FAO. 282 pp.

Karakassis, I., Papageorgiou, N., Kalantzi, I., Sevastou, K., Koutsikopoulos, C., 2013. Adaptation of fish farming production to the environmental characteristics of the receiving marine ecosystems: a proxy to carrying capacity. Aquaculture 408-409:184-190. https://doi.org/10.1016/j.aquaculture.2013.06.002.

Karakassis, I., Tsapakis, M., Hatziyanni, E., Papadopoulou, K.N., Plaiti, W., 2000. Impact of cage farming of fish on the seabed in three Mediterranean coastal areas. ICES Journal of Marine Science, 57: 1462-1471.

Klaoudatos, S.D., Klaoudatos, D.S., Smith, J., Bogdanos, K., Papageorgiou, E., 2006. Assessment of site specific benthic impact of floating cage farming in the eastern Hios island Eastern Aegean Sea Greece. J. Exp. Mar. Biol. Ecol., 338: 96-111.

La Rosa, T., Mirto, S., Favaloro, E., Savona, B., Sara, G., Danovaro, R., Mazzola, A., 2002. Impact on the water column biogeochemistry of a Mediterranean mussel and fish farm. Water Res., 36: 713-721.

Machias, A., Giannoulaki, M., Somarakis, S., Maravelias, C.D., Neofitou, C., Koutsoubas, D., Papadopoulou, K.N., Karakassis, I., 2006. Fish farming effects on local fisheries landings in oligotrophic seas. Aquaculture, 261: 809-816.

Machias, A., Karakassis, I., Labropoulou, M., Somarakis, S., Papadopoulou, K.N., Papaconstantinou, C., 2004. Changes in wild fish assemblages after the establishment of a fish farming zone in an oligotrophic marine ecosystem. Estuar. Coast. Shelf Sci., 60:771-779.

Macias, J.C., Avila Zaragozá, P., Karakassis, I., Sanchez-Jerez, P., Massa, F., Fezzardi, D., Yucel Gier, G., Franičevič, V., Borg, J.A., Chapela Pérez, R.M., Tomassetti, P., Angel, D.L., Marino, G., Nhhala, H., Hamza, H., Carmignac, C., Fourdain, L., 2019. Allocated zones for aquaculture: a guide for the establishment of coastal zones dedicated to aquaculture in the Mediterranean and the Black Sea. In: General fsheries commission for the Mediterranean. Studies and Reviews 97. FAO, Rome, p. 90.

Mayerle, R., Sugama, K., Runte, K-H., Radiarta, N., Maris Vallejo, S., 2017. Spatial Planning of Marine Finfish Aquaculture Facilities in Indonesia. In J. Aguilar-Manjarrez, D. Soto & R. Brummett. Aquaculture zoning, site selection and area management under the ecosystem approach to aquaculture. Full document, pp. 222–252. Report ACS113536. Rome, FAO, and World Bank Group, Washington, DC. 395 pp.

McKindsey, C.W., Thetmeyer, H., Landry, T., Silvert, W., 2006. Review of recent carrying capacity models for bivalve culture and recommendations for research and management. Aquaculture, 261:451-462.

Nordvarg, L., Johansson, T., 2002. The effects of fish farm effluents on the water quality in the Aland archipelago, Baltic Sea. Aquacultural Engineering, 25: 253-279.

Nunes, J.P., Ferreira, J.G., Bricker, S.B., O'Loan, B., Dabrowski, T., Dallaghan, B., Hawkins, A.J.S., O'Connor, B., O'Carroll, T., 2011. Towards an ecosystem approach to aquaculture: assessment of sustainable shellfish cultivation at different scales of space, time and complexity. Aquaculture, 315:369-383.

Pillay, T., 2004. Aquaculture and Environment, Blackwell Publishing.

Pitta, P., Apostolaki, E.T., Giannoulaki, M., Karakassis, I., 2005. Mesoscale changes in the water column in response to fish farming zones in three coastal areas in the Eastern Mediterranean Sea. Estuarine, Coastal and Shelf Science, 65:501-512.

Pitta, P., Karakassis, I., Tsapakis, M., Zivanovic, S., 1999. Natural vs. mariculture induced variability in nutrients and plankton in the Eastern Mediterranean. Hydrobiologia, 391:181-194.

pp. 121–136. FAO/WFT Expert Workshop, 24–28 April 2006, Vancouver, Canada. FAO Fisheries Proceedings No. 10. Rome, FAO. 241 pp. (also available at www.fao.org/docrep/010/a1445e/a1445e00.htm).

Rosenthal, H., Rangeley, R.W., 1988. The effect of a salmon cage culture on the benthic community in a largely enclosed Bay (Dark Harbour, grand Manan Island, N.B., Canada). Fish Health Protection Strategies, 299 pp.

Ross, L.G., Telfer, T.C., Falconer, L., Soto, D., Aguilar-Manjarrez, J., Asmah, R., Bermúdez, J., Beveridge, M.C.M., Byron, C. J., Clément, A., Corner, R., Costa-Pierce, B.A., Cross, S., De Wit, M., Dong, S., Ferreira, J.G., Kapetsky, J.M., Karakassis, I., Leschen, W., Little, D., Lundebye, A.K., Murray, F.J., Phillips, M., Ramos, L., Sadek, S., Scott, P.C., Valle-levinson, A., Waley, D., White, P.G., Zhu, C., 2013a. Carrying capacities and site selection within the ecosystem approach to aquaculture. In: L.G. Ross, T.C. Telfer, L. Falconer, D. Soto & J. Aguilar-Manjarrez, eds. Site selection and carrying capacities for inland and coastal aquaculture, pp. 19–46. FAO/Institute of Aquaculture, University of Stirling, Expert Workshop, 6–8 December 2010. Stirling, the United Kingdom of Great Britain and Northern Ireland. FAO Fisheries and Aquaculture Proceedings No. 21. Rome, FAO. 282 pp.

Ross, L.G., Telfer, T.C., Falconer, L., Soto, D., Aguilar-Manjarrez, J., 2013b. Site selection and carrying capacities for inland and coastal aquaculture. FAO/Institute of Aquaculture, University of Stirling, Expert Workshop, 6-8 December 2010. Stirling, the United Kingdom of Great Britain and Northern Ireland. FAO Fisheries and Aquaculture Proceedings No. 21. Rome, FAO. 46 pp.

Saurel, C., Ferreira, J.G., Cheney, D., Suhrbier, A., Dewey, B., Davis, J., Cordell, J., 2014. Ecosystem goods and services from Manila clam culture in Puget Sound: a modelling analysis.

Smith, C., Machias, A., Giannoulaki, M., Somarakis, S., Papadopoulou, K.N., Karakassis, I., 2003. Diversity study of wild fish fauna aggregating around fish farm cages by means of remotely operated vehicle (ROV). Abstr 7th Hel Symp Oceanogr & Fish p 227.

Soto, D., Norambuena, F., 2004. Evaluation of salmon farming effects on marine systems in the inner seas of southern Chile: a large-scale mensurative experiment. J Appl Ichtyol., 20:493-501.

Soto, D., Salazar, F.J., Alfaro, M.A., 2007. Considerations for comparative evaluation of environmental costs of livestock and salmon farming in southern Chile. In D.M. Bartley, C. Brugère, D. Soto, P. Gerber & B. Harvey, eds. Comparative assessment of the environmental costs of aquaculture and other food production sectors: methods for meaningful comparisons, pp. 121–136. FAO/WFT Expert Workshop, 24–28 April 2006, Vancouver, Canada. FAO Fisheries Proceedings No. 10. Rome, FAO. 241 pp. (also available at www.fao.org/docrep/010/a1445e/a1445e00.htm).

Stelzenmüller, V., Gimpel, A., Gopnik, M., Gee, K., 2017. Aquaculture site-selection and marine spatial planning: the roles of GIS-based tools and models. In Aquaculture Perspective of Multi-Use Sites in the Open Ocean, pp. 131-148. Springer International Publishing, Cham. http://link.springer.com/10.1007/978-3-319-51159-7_6.

Stigebrandt, A., 2011. Carrying capacity: general principles of model construction. Aquaculture

Syandri, H., Azrita, Niagara, 2016. Trophic status and load capacity of water Pollution waste fish culture with floating net cages in Maninjau Lake, Indonesia. Ecology, Environment and Conservation Paper 22(1):469-476.

Tett, P., Portilla, E., Gillibrand, P.A., Inall, M.E., 2011. Carrying and assimilative capacities: the ACExR-LESV model for sea-loch aquaculture. Aquaculture Research, 42: 51–67. doi:10.1111/j.1365-2109.2010.02729.x.

Tomassetti, P., Porrello, S., 2005. Polychaetes as indicators of marine fish farm organic enrichment. Aquacult. Int., 13:109-128.

Tuya, F., Sanchez-Jerez, P., Dempster, T., Boyra, A., Haroun, R.J., 2006 Changes in demersal wild fish aggregations beneath a sea-cage fish farm after the cessation of farming. J. Fish. Biol., 69: 682-697.

UN-WPP (United Nations Department of Economic and Social Afairs, Population Division), 2024. In: World population prospects 2024: summary of results UN DESA/POP/2024/TR/NO. 3. https://population.un.org/wpp/Download/Standard/MostUsed (Accessed 01 September 2024).

Vega Fernandez, T., D’Anna, G., Badalamenti, F., Pipitone, C., Coppola, M., Rivas, G., Modica, A., 2003. Fish fauna associated to an off-shore aquaculture system in the Gulf of Castellammare (NW Sicily). Biol. Mar. Mediterr., 10:755-759.

Vita, R., Marin, A., Jimenez-Brinquis, B., Cesar, A., Marin-Guirao, L., Borredat, M., 2004. Aquaculture of Bluefin tuna in the Mediterranean: evaluation of organic particulate wastes. Aquacult. Res., 35:1384-1387.

Vollenweider, R.A., 1968. Scientific fundamentals of the eutrophication of lakes and flowing water with particular reference to nitrogen and phosphorus as factors in eutrophication. Technical Report DASISU/68–27. Paris, OECD.

Weitzman, J., Filgueira, R., 2020. The evolution and application of carrying capacity in aquaculture: Towards a research agenda. Rev. Aquac. 12:1297-1322. https://doi.org/10.1111/raq.12383.

Weitzman, J., Filgueira, R., Grant, J., 2021. Development of best practices for more holistic assessments of carrying capacity of aquaculture. J. Environ. Manag. 287:112278. https://doi.org/10.1016/j.jenvman.2021.112278.

Weitzman, J., Steeves, L., Bradford, J., Filgueira, R., 2019. Far-field and near-field effects of marine aquaculture. In: C Shepperd (ed.) World Seas: An Environmental Evaluation Vol III Ecological Issues and Environmental Impacts, 2nd edn, pp. 197-220. Academic Press, London, UK.

Yıldırım, Ö., Korkut, Y., 2004. Effect of Aquafeeds on the Environment. Ege University Journal of Fisheries & Aquatic Sciences, 21(1-2), 167-172.

Yigit, M., Ergun, S., Buyukates, Y., Ates, A.S., Ozdilek, H.G., 2021. Physical carrying capacity of a potential aquaculture site in the Mediterranean: the case of Sigacik Bay, Turkey. Environ. Sci, Pollut, Res. 28:9753-9759. https://doi.org/10.1007/s11356-020-11455-y.

Yigit, M., Ergün, S., Buyukates, Y., Ates, A.S., Ozdilek, H.G., Acar, S., 2024. Assessment of physical carrying capacity of a mariculture zone designated in the Aegean Sea, Aquaculture International, 32:2249-2261, https://doi.org/10.1007/s10499-023-01268-4.

Yucel-Gier, G., 2017. Mariculture Parks in Turkey. In J. Aguilar-Manjarrez, D. Soto & R. Brummett. Aquaculture zoning, site selection and area management under the ecosystem approach to aquaculture. Full document, pp. 314–331. Report ACS113536. Rome, FAO, and World Bank Group, Washington, DC. 395 pp.

Yucel-Gier, G., Kucuksezgin, F., Kocak, F., 2007. Effects of fish farming on nutrients and benthic community structure in the Eastern Aegean (Turkey). Aquacult. Res., 38: 256-267.

Yuningsih, H.D., Soedarsono, P., Anggoro, S., 2014. The relationship of organic matters to water productivity at Rawa Pening lake. Diponegoro Journal of Maquares 3(1):37-43.

Zhu, C., Dong, S., 2013. Aquaculture site selection and carrying capacity management in the People's Republic of China. In L.G. Ross, T.C. Telfer, L. Falconer, D. Soto & J. Aguilar Manjarrez, eds. Site selection and carrying capacities for inland and coastal aquaculture, pp. 219-230. FAO/Institute of Aquaculture, University of Stirling, Expert Workshop, 6-8 December

2010. Stirling, the United Kingdom of Great Britain and Northern Ireland. FAO Fisheries and Aquaculture Proceedings No. 21. Rome, FAO. 282 pp.

CHAPTER 2

THE DEVELOPMENT OF WATER QUALITY INDEX (WQI): A REVIEW

Özgür CANPOLAT[1]*, Suel ÖZTÜRK*[2]

---

[1] Prof. Dr. Fırat University, Fisheries Faculty, Elazığ ORCID: 0000-0001-7498-600X

[2] Fırat University, Fisheries Faculty, Elazığ ORCID: 0009-0000-0699-1085

INTRODUCTION

Water and water resources are not only very important for all living organisms due to the many nutrients and minerals they contain, but are also invaluable natural resources with social and economic value for humans (Vasistha and Ganguly, 2020; Akhtar et al., 2021).

Water quality (WQ) is affected by both human activities and natural processes. WQ is generally defined as the physical, chemical and biological properties of water for its suitability for a particular use (Johnson et al., 1997). While WQ in aquatic ecosystems depends on natural factors (such as hydrological, atmospheric, climatic, topographic and lithological factors), anthropogenic activities (industrial practices, irregular urbanization and agricultural activities, etc.) have a significant effect on WQ (Fig. 1), (Vasistha and Ganguly, 2020; Akhtar et al., 2021). Every year, 300 to 400 million tons of toxic substances are discharged into aquatic ecosystems. In developing countries, 80% of sewage is discharged directly into aquatic ecosystem without treatment (WHO-UN, 2010).

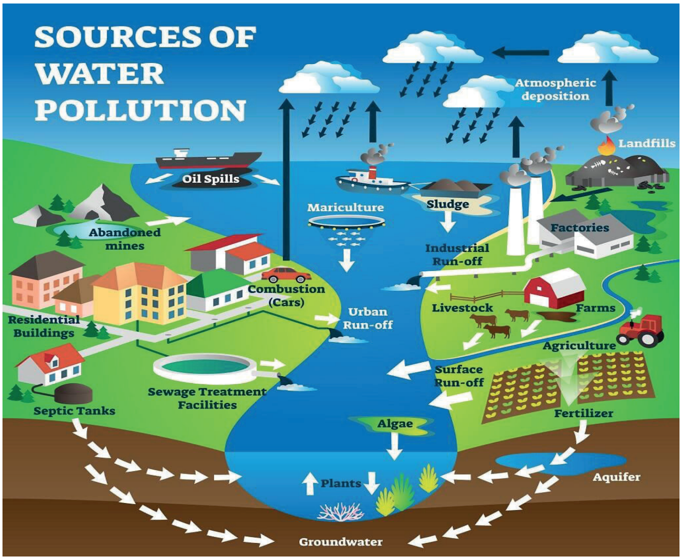

Figure 1. *Sources of water pollution (https://consciouswater.ca/what-is-water-pollution/)*

The concept of WQ was first used in Germany in 1848 to categorize water according to its purity and degree of pollution (Lumb et al., 2011). Many local and international organizations have tried to ensure acceptable WQ by preparing guidelines and criteria regarding the concentration levels of parameters in water bodies (Al Yousif and Chabuk, 2023). The assessment of WQ is usually carried out by determining the physico-chemical and biological properties or parameters of water according to a set of standards (Chapman, 1996). As a result, the World Health Organization (WHO) established water quality guidelines with the aim of following the changes in physical, chemical and biological parameters that are affected by many external and internal factors (WHO, 2017). These data are used to determine whether the water is suitable for consumption or safe for the environment (Chapman, 1996).

WQ standards refer to expressions and numerical values that fall into three components that define WQ. These components are listed below;

a) Identified uses of the water body depending on the purpose of use of water (aquatic life, water supply, agriculture and recreation),

b) WQ criteria and general expressions for various parameters,

c) To ensure the sustainable use of each water body and to take protection measures (Adelagun et al., 2021).

Pollutants found in water have direct or indirect negative effects on aquatic ecosystems. The effects of substances that may be found in drinking water on human health are determined by the type and amount of these substances. While each country determines its own drinking water criteria, international organizations such as the European Union (EU) and the World Health Organization (WHO) have also set limit values, especially for harmful chemical and biological substances (Lirika et al., 2013). In addition, monitoring the quality of water bodies has been made mandatory for EU member states within the scope of the Water Framework Directive (WFD): European Union, and the parameters to be monitored are included in this directive (WHO, 2011). However, evaluating and interpreting these parameters one by one is a very difficult and time-consuming process for both regulatory bodies and experts working on this subject. Therefore, recently, many studies have focused on expressing WQ in a more practical, comprehensive, understandable and comparable way (Akkoyunlu, 2012).

In recent years, due to increased pollution, strict regulations and control over the monitoring of surface water bodies for sustainable use have increased rapidly (Vasisthan and Ganguly, 2020). In parallel with the increasing importance of water quality indices (WQI), the number of scientific studies in this field is also increasing. According to Scopus, when the number of studies conducted on WQI from 1978 to 2022 is examined in general, it is reported that the number of studies has increased over time, especially in recent years. Accordingly, while the number of studies increased from 1 to 13 between 1975 and 1988, this number increased to 46 studies in 1998 and gradually reached 466 publications in 2011. Research on WQI has increased significantly in the last decade and reached its highest value with 1316 studies in 2022. Considering these values, it shows that WQI studies have become an important research topic. Considering the development of WQI research by country from 1975 to 2022, the first three countries according to Scopus were China, India and the USA with 2356, 1678 and 1241 studies, respectively. Iran ranked fourth with 409 studies, Brazil ranked fifth with 375 studies and Italy ranked sixth with 336 studies. There are approximately the same number of studies in Malaysia and Spain, 321 and 320 respectively. 303 studies were conducted in Spain and 210 studies were conducted in Turkey (Scopus, 2022). These numerical data show that developing countries like India, despite not having strong economic power, advanced technology and world-class scientific research team, attach great importance to the protection of WQ. Because WQ is so important for the long-term social and economic development of these nations (Zhang, 2019).

In this study, general information will be given about the WQI, its historical development and some commonly WQI.

What is Water Quality Index (WQI)

Generally, the dimensionless number that combines various parameters to evaluate the WQ of any water body is called WQI (Al Yousif and Chabuk, 2023). WQI methods allow to significantly reduce the amount of data and describe the status of WQ with a single number (Kachroud et al., 2019; Al Yousif and Chabuk, 2023).

One of the most commonly used tools to describe WQ is WQI. WQI are based on physical, chemical and biological factors that are combined into a single value ranging from 0 to 100. WQI involve four processes: parameter selection, transformation of the raw data to a common scale, providing weights and aggregation of sub-index values.

Historical Development of Water Quality Index (WQI)

Quality indices, whose primitive forms date back about 150 years and first appeared in Germany in 1848, have become widespread in the last thirty years. The presence or absence of certain organisms in water was used as indicators of water resources. Since then, different systems have been developed in various European countries to classify the quality of water. Water classification systems are generally of two types. It is related to the amount of pollution and microscopic and macroscopic communities. Rather than providing a numerical value to indicate WQ, these classification systems categorize various pollution classes within water bodies. On the other hand, indices that use a numerical measure for the degrees shown in WQ levels have been used very frequently recently, and the first WQI was proposed by Horton in 1960, who attached great importance to the development of the "Water Quality Index" method in order to simplify WQ data (Abbasi, 2002; Liou et al., 2004).

Horton (1965) initially developed a system for rating WQ through index numbers and offered a tool for reducing water pollution since the terms "water quality" and "pollution" are interrelated (Chidiac et al., 2023). The WQI developed by Horton took into account the 10 most commonly used WQ variables in the United States, such as dissolved oxygen (DO), pH, coliforms, specific conductance, alkalinity and chloride, etc., and has been widely applied and used. Horton (1965) selected these parameters in his study. He gave rating scales to create sub-indices ranging from 0 to 100, where the highest quality rating provided was 100. The weightage assigned to different parameters varied from 1 to 4 (Horton, 1965). This index is accepted in European, African and Asian countries (Chidiac et al., 2023).

Horton identified the steps to be followed in developing an index as follows; (i) determining the quality characteristics on which the index is to be based; (ii) creating a rating scale for each characteristic and (iii) weighting of the some characteristics (Kachroud et al., 2019).

The original Horton model used seven physicochemical parameters (dissolved oxygen, pH, coliforms, electrical conductivity (EC), carbon chloroforms extract, alkalinity, and chlorides) of WQ (Abbasi and Abbasi, 2012; Shah and Joshi, 2015; Uddin et al., 2021). Electrical conductivity is designed to help approximate total dissolved solids and carbon chloroform extract (CCE) reflects the influence of organic matter. Index weightings range from 1 to 4. Horton's index, in particular, does not include any toxic chemicals (Table 1). This method, which has been widely used to classify water resources according to their purity level, uses the weighted arithmetic average WQI method (Rana and Ganguly, 1920).

In the Horton model, five WQ range need to be considered to determine the final WQI value (Table 2), (Uddin et al., 2021).

Table 1. *Sub-index ranges and weightings of the index used by Horton (1965)*

Sub-index	Parameter						
	DO (%)	**Coliform (number/100 mL)**	**CCE (0,0001 mg/L)**	**pH**	**Cl (mg/L)**	**EC(µmho/ cm)**	**Alkalinity (mg/L)**
100	>70	<1	0-100	6-8	0-100	0-750	20-100
80	70-50	1-5	100-200	5-6;8-9	100-175	750-1500	5-20;100-200
60	50-30	5-10	200-300				
40				4-5;9-10	175-250	1500-2500	0-5;>200
30	30-10	10-20	300-400				
0	<10	>20	>400	<4;>10	>250	>2500	Asit
Weight Degrees	4	2	1	4	1	1	1

Table 2. *WQ range considered in the Horton model (Horton, 1965)*

WQ range	Rating of WQI
91–100	Very good
71 – 90	Good
51 – 70	Poor
Bad	31 – 50
Very bad	0 – 30

The following formula is used to calculate the WQI:

$$WQI = (\sum S_n x W_n \text{ x } m_1 \text{ x } m_2) / \sum W_n$$

where S_n is the sub-index assigned to the nth variable

W_n is the relative weight of the nth variable

m_1 is a temperature correction factor (0.5 if the temperature is below 34 °C, else1)

m_2 is a correction pollution factor (0.5 or 1).

The categorization of water determined from the WQIs varies between 0 and 100 according to its relative impact (Table 3).

Table 3. *Rating of WQ as per Arithmetic Average Index Method (Horton, 1965)*

WQI range	Rating of WQ	Grade
0–25	Excellent quality	A
26–50	Good quality	B
51–75	Poor quality	C
76–100	Very poor quality	D
Above 100	Highly unsuitable	E

Brown et al. (1970) established a new WQI containing nine parameters (temperature, pH, Biochemical Oxygen Demand (BOD), dissolved oxygen (DO), fecal coliform (FC), total phosphate and nitrate concentrations, turbidity, and total solid content). There are five classes (WQ: red-very poor, orange-poor, yellow-average, green-good) based on the opinion of 142 researchers who are experts in the field of WQ (Brown et al. 1970, 1973; Bharti and Katyal 2011; Rana and Ganguly, 2020).

Brown et al. (1970) used arithmetic addition in the initial index, but later determined that geometric addition was better and more sensitive than arithmetic addition when a single variable exceeded the norm. (Lumb et al., 2011; Kachroud et al., 2019). This index is called NSFWQI because these studies were supported by the National Sanitation Foundation (Kachroud et al., 2019).

In Europe, Prati et al. (1971) proposed an index based on WQ standards in which pollution levels are determined by the concentrations of pollutants. Thirteen parameters (pH, Biological Oxygen Demand (BOD), Chemical Oxygen Demand (COD), Dissolved Oxygen (DO), concentrations of permanganate, ammonium, nitrate, chloride, iron, manganese, Alkyl Benzene sulphonates, suspended solids (SS), Carbon Chloroform Extract) are used in this WQI (Prati et al 1971).

Nemerow and Sumitomo (1971) proposed WQI having three specific uses which when combined form an overall WQI. Bhargava (1983a) proposed a new water quality index in India which showed that the pollution load is more specific in the combination of variables (Al Yousif and Chabuk, 2023).

Deininger and Landwehr (1971) proposed a new water quality index that is conceptually similar to the index of Brown et al. (1972). This index includes 12 variables for surface water (temperature, pH, turbidity, colour and hardness Dissolved Oxygen (DO), fecal coliform (FC), Biological Oxygen Demand (BOD), the concentrations of nitrate, phosphate, phenol, dissolved solid) and 14 variables for groundwater (the same variables given for surface waters plus iron and fluoride concentrations) (Kachroud et al., 2019).

Another model similar to the WQI model developed by Horton was developed by Dinius (1972). This model was developed to evaluate the cost of remediation in case of pollution of water resources, and this model uses a decreasing category scale from 100 to 0, where 0 represents very poor quality and 100 represents excellent WQ (Dinius 1972,1987).

Tiwari and Mishra (1985) developed a new model by making only a slight change in the weighting method from the basic principles of Horton (1965). In this method, normative values of main variables of water resources are used instead of previously given methodologies (Tiwari and Mishra, 1985). The classification of water quality used in this method is given in Table 4.

Table 4. *Classification of WQ (Tiwari and Mishra 1985)*

WQI Quality	Range
< 26	Excellent
26–50	Good
51–75	Medium
76–100	Poor
> 100	Unsuitable

The emergence of new indices in the twenty-first century has led to a significant simplification of the currently used formulas and the definition of the index's field of application. An example of this is the evaluation of the General Pollution Index depending on various WQ variables, based on the measurement and classification of each variable (Al Yousif and Chabuk, 2023).

The first WQI used weighted average (arithmetic means) techniques, and later geometric aggregations were used with certain modifications in the calculations. Since this attempt was supported by the National Sanitation Foundation (NSF), this index was named the National Sanitation Foundation Water Quality Index (NSF-WQI).

The mathematical expression is gives as:

$$WQI= \Sigma^{n}_{i=1} q_i W_i$$

where q_i is the quality class for the n^{th} variable

W_i is the relative weight for the n^{th} variable ($W_i = 1$)

The detailed ratings of WQ for NSFWQI is as explained in Table 5.

Table 5. *Weights for variables (Brown et al., 1970, 1973)*

Variables	Weight
Dissolved solids	0.07
DO	0.17
BOD	0.11
Nitrats	0.10
pH	0.11
Phosphates	0.10
Temperature	0.10
Turbidity	0.08
Fecal coliform	0.16
Total	1.00

Many researchers (House and Newsome 1989; Sargaonkar and Deshpande 2003; Icaga 2007; Silvert 2000; Sharma et al. 2014; Yadav et al., 2010; Ramakrishnaiah et al., 2009; CCME, 2001; Bhargava, 1983a,b; Balan et al., 2012; Shah and Joshi, 2015) have developed different WQI in accordance with the development principles of NSFWQI.

However, numerous WQI have been formulated by many national and international organizations, such as the National Sanitation Foundation Water Quality Index (NSFWQI), the Canadian Council of Ministers of the Environment Water Quality Index (CCMEWQI), the Oregon Water Quality Index (OWQI), the Weighted Arithmetic Water Quality Index (WAWQI), etc. These WQI have been applied to assess WQ in a particular area (Lumb et al., 2002; Chaturvedi and Bassin, 2010). In addition, these WQIs are often based on a varying number and type of WQ parameters compared to the relevant standards of a particular area. WQIs are accredited to effectively and timely display annual cycles in WQ, spatial and temporal changes in WQ, and trends in WQ even at low concentrations (Tyagi et al., 2013).

Commonly Used Water Quality Indices

1. National Sanitation Foundation Water Quality Index (NSFWQI)

The United States National Sanitation Foundation (NSF) developed the National Sanitation Foundation Water Quality Index (NSFWQI) in 1970 (Singh et al., 2013; Samadi et al., 2015). This WQI, which is used to calculate and evaluate the water quality index of many water bodies, has been extensively field-tested (Singh et al., 2013). However, this index belongs to the group of publicly available indices and represents a general water quality. However, it does not take into account the water usage capacity (Bharti and Katyal, 2011; Ewaid, 2017). NSFWQI is an index based on the analysis of nine variables or parameters (BOD, dissolved oxygen, nitrate (NO_3), Total Phosphate PO_4, temperature, turbidity, Total Solids

(TS), Fecal Coliform (FC) and pH) and is widely applied and accepted in Asian, African and European countries (Tyagi et al., 2013; Chidiac et al., 2023).

The formula for calculating the index is as follows:

$$NSF - WQI = \sum P_i C_i$$

Here;

C represents the normalized subindex values

P is the weight factor ranging from 0 to 1

i represents the parameter

n is the total number of parameters considered in the process (Tyagi et al., 2013).

In this method, the rating ranges from 0 to 100 (Table 6), with 100 representing excellent WQ conditions and zero indicating water that is unfit for use and requires further treatment (Samadi et al., 2015).

Table 6. *Colors and defnition used in the classifcation of pollution using NSF-WQI (Roozbahani and Boldaji, 2013)*

Color	**The numerical value index**	**Definition**
Red	0–25	Very bad
Orange	26–50	Bad
Yellow	51–70	Moderate
Green	71–90	Good
Blue	91–100	Excellent

In this method; the calculated values are related to the potential use of water. The disadvantage of this method; data loss occurs during the calculation, it represents the general WQ, not the specific use of water, there is uncertainty in complex environmental issues (Roozbahani and Boldaji, 2013).

2. British Columbia Water Quality Index (BCWQI)

The British Columbia Water Quality Index was developed by the Canadian Ministry of Environment in 1995. This index is similar to the CCMEWQI, where WQ parameters are measured and non-compliant measurements are determined by comparing them with a predetermined limit. It provides a classification based on all available measurement parameters. The following equation is used to calculate the index value:

$$BCWQI = 100 - \left(\frac{\sqrt{F_1^2 + F_2^2 + (F_3/3)^2}}{1.453} \right)$$

The number 1.453 was chosen to ensure that the index number of the scale is from zero to 100. Repeated sampling and increasing stations increase the accuracy of the BCWQI. The disadvantage of this method is that this index does not show the trend of WQ until it deviates from its standard limit (Suryawnshi et al., 2018; Scopus, 2022).

3. Canadian Council of Ministers of the Environment Water Quality Index (CCMEWQI)

The Canadian Council of Ministers of the Environment Water Quality Index (CCMEWQI); The Canadian Water Quality Index adopted the conceptual model of the British Columbia Water Quality Index (BCWQI) based on relative sub-indices (Paun et al., 2016; Kizar, 2018). The Canadian Water Quality Index consists of three elements: extent (number of variables that do not meet WQ targets), frequency (number of times these targets are met) and amplitude (values that do not meet targets). The CCMEWQI is used to assess water quality in specific monitoring areas in Canada to determine the suitability of water bodies to support aquatic life (Paun e al., 2016). In addition, this index, which provides WQ information for both the administration and the public, is applied with some minor changes in many water-related institutions in many countries (Tyagi et al., 2013). Simplifying complex and technical data, the CCMEWQI method tests multivariate WQ data and compares the data to user-specified benchmarks (Tirkey et al., 2015). This method requires at least four parameters and sampling at least four times. However, the user decides which parameters to select in this index (Uddin et al., 2021). In addition, the parameters may vary from one station to another (Tyagi et al., 2013). The values obtained vary between 0 and 100. Here, 0 represents poor WQ and 100 represents excellent WQ (Table 7), (Chidiac et al., 2023).

Table 7. *The CCME model proposed four WQ classes (Uddin et al., 2021)*

WQ Classes	WQI value	WQ	Definition
I	95-100	Excellent	Natural WQ
II	80-94	Good	WQ is departed from natural or desirable levels
III	65-79	Fair	WQ condition sometimes departs from natural or desirable levels
IV	45-64	Marginal	WQ is frequently threatened or impaired; conditions often depart from natural or desirable level
V	0-44	Poor	WQ is not suitable for using purposes at any level

The aggregation function used in other WQI is quite different from the one used by CCME. The aggregation function used by CCME is given below;

$$WQI = 100 - \frac{\sqrt{F_1^{\,2} + F_2^{\,2} + F_3^{\,2}}}{1.732}$$

Conceptually, the CCMEWQI includes three factors (CCME, 2001).

(a) F_1 is the percentage of total parameters that do not meet the specified objectives and is called 'coverage'

The formula is given below.

$$F_1 = \left[\frac{number\ of\ failed\ parameters}{total\ number\ of\ parameters}\right] x100$$

(b) F_2 is the percentage of individual test values that do not reach their objectives values (failed tests) and is called the 'frequency'.

The formula is given below.

$$F_2 = \left[\frac{number\ of\ failed\ tests}{total\ number\ of\ tests}\right] x100$$

(c) F_3 is a measure of the amount by which the est values fail to meet their objectives, called the 'amplitude'. The amplitude is calculated by an asymptotic function that scales the normalized sum of the excursions (nse) of the test values from the objectives to yield a value between 0 and 100 using:

$$F_3 = \left[\frac{nse}{0.01(nse) + 0.01}\right]$$

For the test value falling below the objective value, the excursion is calculated according to the formula given below:

$$excursion_i = \left[\frac{failed\ test\ value_i}{Objective_j}\right] - 1$$

If the test value is above the objective value, the excursion value is calculated according to the formula given below:

$$excursion_i = \left[\frac{Objective_j}{failed\ test\ value_i}\right] - 1$$

nse represents the total amount by which individual test values are out of compliance. **nse** is the ratio of the sum of the deviations of individual tests from their targets to the total number of tests (both meeting and not meeting targets). Its mathematical calculation is given below.

$$nse = \left[\frac{\sum_{i=1}^{n} excursion_j}{total\ number\ of\ test}\right] - 1$$

The coefficient 1.732 specified in the equation is used as a normalization factor to ensure that the resulting water quality index is in the range of 0 to 100. Where 0 indicates "worst" and 100 indicates "best" WQ. The factor of 1.732 arises because each of the three individual index factors (F_1, F_2 and F_3) can have a maximum value of 100 giving a maximum value for the numerator of 173.2 (Uddin et al., 2021).

4. Oregon Water Quality Index (OWQI)

The OWQI was developed in the 1970s by the Oregon Department of the Environment to summarize and evaluate WQ status and trends for WQ condition assessment reports (Cude, 2001). The OWQI is a single number that generates a score to assess the WQ of the Oregon River and is applicable to other geographic regions (Tyagi et al., 2013; Singh et al., 2013). A variant of NSFWQI, OWQI, used to assess water quality for swimming and fishing and to manage large streams, has been widely accepted and implemented in Oregon (USA) and Idaho (USA), (Sutadian et al., 2016; Lumb et al., 2011). WQ science has advanced significantly since the inception of the OWQI in 1970. Researchers developing water quality indices have benefited from the increased understanding of stream functionality since 1978 (Bharti and Katyal, 2011). OWQI, a type of water classification based on consumption type and application (drinking, industrial, etc.), belongs to a group of specific consumption indices (Shah and Joshi, 2015). The original OWQI was dropped in 1983 due to the excessive resources required to calculate and report the results. However, the need for accessible WQ information and improvements in the availability of software and computer hardware have led to renewed interest in the index (Cude, 2001). As a result of advances in computer technology, improved data display and visualization tools, and a better understanding of water quality, researchers updated the OWQI in 1995 by refining the original subindex, adding temperature and total phosphorus subindexes, and improving clustering (Tirkey et al., 2015).

OWQI calculations require eight variables: temperature, dissolved oxygen (% saturation and concentration), BOD, pH, total solids, ammonia, nitrate nitrite, total phosphorus and bacteria. The data obtained as a result of the calculations in OWQI are expressed with values ranging from 10 to 100. Here, 10 represents the worst water quality and 100 represents the most ideal water quality (Table 8), (Brown, 2019).

The OWQI is calculated as follows;

$$QWQI = \sqrt{\frac{n}{\sum_{i=1}^{n} \frac{1}{SI_i^2}}}$$

Where, SIi = Sub-index of each parameter, n = Number of sub-index

Table 8. *OWQI quality classification (Tyagi et al., 2013)*

WQI Value	**Water Quality Rating**
90-100	Excellent water quality
85-89	Good water quality
80-84	Fair water quality
60-79	Poor water quality
0-59	Very poor water quality

This method recognizes that various WQ parameters have different importance on WQ at different times and places. The formula used in this method is sensitive to changing conditions and their effects on WQ. Therefore, this method assumes that various WQ parameters have different importance on WQ at different times and places (AWDO, 2018).

5. Weighted Arithmetic Water Quality Index (WAWQI)

Weighted Arithmetic Water Quality Index (WAWQI) is used to calculate the purified WQI, in other words, this method classifies WQ according to the degree of purity using the most frequently measured WQ variables (Table 9), (Paun et al., 2016; Kizar, 2018). This index is widely used by scientists (Tyagi et al., 2013; Singh et al., 2013).

The calculation method is as follows.

$WQI = \sum WiQi / \sum Wi$

The quality grade (Qi) for each parameter is calculated using this equation:

$Qi = 100[(Ve - Vi / Vs - Vi)]$

Here;

Ve= experimental value

Vi= ideal value (pH=7 and Dissolved oxygen = 14.6 mg/L),

Vs= standard values,

$Wi = K / Vs$,

Wi= unit weight for each parameter,

K= proportionality constant = $1/\sum(1/Vs)$

Table 9. *WAWQI and status of WQ (Yogendra and Puttaiah, 2008)*

WQI level	WQ status
0-25	Excellent WQ
26-50	Good WQ
51-75	Poor WQ
76-100	Very poor WQ
>100	Unsuitable for drinking

The advantages of this method are that it incorporates multiple WQ parameters into one mathematical equation. Fewer parameters are required compared to all WQ parameters for a specific use. It is useful for communicating general WQ information to concerned citizens and policy makers (Shah and Joshi, 2015).

A summary of the structures of the most common WQI models is given in Table 10.

Table 10. *Summary of structures of most common WQI models (Uddin et al., 2021)*

WQI model	**Model Components**			
	No of parameters and selection process	**Sub-indexing procedure**	**Parameter Weighting**	**Rating scale**
Horton index (1960)[a]	•8 parameters suggested • parameters significance and data availability	•parameters value used as sub-index value, and subindex ranges from 0 to 100 assigned	•fixed and unequal system (4 for DO and 1 for other parameters) suggested	•Five categories - Very good (91–100) - Good (71–90) - Poor (51–70) - Bad (31–50) - Very bad (0–30)
NSF index (1965)[b]	•11 parameters • Used Delphi technique	• used water quality standard guideline and scale ranged from 0 to 1; When, Parameter value < standard = 1, Parameter value > standard = 0 modified	•the expert panel judgement, and sum of weight value is equal to 1 given	Five categories - excellent (90–100) - good (70–89) - medium (50–69) - bad (25–49) - very Bad (0–24)
SRDD Index (1970)[c]	•10 parameters • Used Delph	• Used expert opinion, and it ranged from 0 to 100 recommended by SRDD	•panel based and sum of weight value equal to 1 recommended by SRDD	seven classification - clean (90–100) - good (80–89) - good with treatment (70–79) - tolerable (40–69) - polluted (30–39) - several polluted (20–29) - piggery waste (0–19)
Dinius index (1972)[d] *modified version of NSF index	• 11 parameters • Delphi technique	• parameters value directly assigned as sub-index value	•used unequal weight • sum of Weighting value is equal to 10	Five classification - Purification not required (90–100) - minor purification required (80–90) - treatment required (50–80) - doutful (40–50)

Ross Index (1977)[e]	• 4 general WQ parameters • Delphi method	•Expert panel judgement based sub-index system	•expert based and sum of weight value is equal to 1 given	Not specified
Bascaron Index (1979)[f]	• 26 parameters were suggested	•Parameters value directly transformed into subindex value using liner transformation function • It ranges from 0 to 100	•Used unequal and fixed weighting technique • ranges from 1 to 4 • Sum of weight value is equal to 54	Five classes - Excellent (90– 100) - Good (70–90) - Medium (50–70) - Bad (25–50) - Very bad (0–25)
Oregon Index (1980)[g] *refined version of NSF index	• 8 parameters used Delphi process	•Logarithmic transformation and nonliner regression were used for generating sub-index	•Sub-index values directly used as Weighting factors	• Five classes - excellent (90–100) - good (85–89) - fair (80–84) - poor (60–79) - very poor (<60)
EQ index (1982)[h]	•9 parameters recommended • Adopted Delphi method	•Fixed system, and used national-international water quality guideline • Used expert opinion	• fixed and unequal (0.1 for physical, chemical and biological parameters, and 0.15 for organic and inorganic r parameters)	• Five categories - excellent (90–100) - very good (80–89) - good (70–79) - fair (55–69) - poor (<55)

WQI model	**Model Components**			
	No of parameters and selection process	**Sub-indexing procedure**	**Parameter Weighting**	**Rating scale**
House index (1986)[i] *refined version of SRDD index	• 9 parameters • Key personnel interview • Expert panel judgement process	• Parameters value directly used as a sub-index • Sub-index scale ranges from 10 to 100	• the expert panel judgement, and sum of weight value is equal to 1	• recommended 4 classification - high quality (71–100) - reasonable quality (51–70) - moderate quality (31–50) - polluted (10–30)
Smith Index (1990)[j]	• 7 parameters • Used Delphi technique	• Fixed system and expert based	• Not required	•Not specified
Dojildo Index (1994)[k]	•26 parameters •Open (additional group) and close system (basic parameters group)	• Not required	• Not required	• Four quality recommended by Dojildo - Very clean (75 – 100) - clean (50–75) - polluted (25–50) - very polluted (0–25)
British Colombia Index (1995)[l]	•Used common monitoring parameters • Open choice system • At least 10 parameters	•Sub-index assigned based on expert opinion	•Unequal and expert based	• Five classes - excellent (0–3) - good (4–17) - fair (18–43) - borderline (44–59) - poor (60–100)
Dalmatian Index (1999)[m]*modified version of SRDD index	•8 parameters • Delphi technique	• Parameters value used directly as sub-index	• Fixed and unequal weight fixed by expert panel • Sum of weight value equal to 1.	•Categories not specified
CCME (2001)[n] * reformed version of BCWQI index	• 4 WQ parameters • Delphi technique	• Not required	• Not required	• Suggested 5 types of WQ - excellent (95 – 100) - Good (80 – 94) - fair (65 – 79) - marginal (45 – 65) - poor (0 – 44)

Liou Index (2004)[o]	• 13 parameters were used • Parameters were selected based on environmental and health significance	• Parameters actual concentration directly used as sub-index	• Equal Weighting system • Weighting factors were generated by the using rating curves that were illustrated based on the standard guideline of WQ variables	•Not specified

WQI model	**Model Components**			
	No of parameters and selection process	**Sub-indexing procedure**	**Parameter Weighting**	**Rating scale**
Said Index (2004)[p]	• 5 parameters • Based on environmental significance	• Parameters value used as sub-indexa	•mathematical function (Eq. (8))	•Three WQ classification and index value ranges from 0 to 3. - highest purity (3) - marginal quality (<1)
Malaysian Index (2007)[q]	• 6 parameters used	•• Unequal and close system • Expert based • Sum of weight is 1	•Simple additive function used	•Parameter based individual rating scale used
Hanh Index (2010)[r]	• 8 parameters, • Based on monitoring data availability	•Rating curve-based sunindexing system • curve developed based on Vietnamese surface water quality standards	• not required	• five quality classification - Excellent (91–100) - good (76–90) - fair (51–75) - marginal (26–50) - poor (<25)
Almeida Index (2012)[s]	• 10 WQ parameters • Delphi technique	• Rating curve-based sunindexing system • Parameters rating curve recommended by expert panel	• Close and unequal system • Weighting factors fixed by expert panel • Sum of weight value is 1	• Four categories - Excellent (91–100) - good (81–90) - medium (71–80) - poor (<25) - poor (<70)

West Java Index (2017)[t]	• 13 parameters • Parameters were selected based on monitoring data availability and comparison of standards.	• Used straightforward mathematical function • Adopted guideline value for generating subindexing	• Multi decision making tools like as Analytic Hierarchy Process (AHP). • Fixed and unequal weight values • Expert based opinion • The sum of weight values is equal to 1	• Five classification - Excellent (90–100) - good (90–75) - Fair (75–50) - Marginal (50–25) - poor (25–5)

Indices application Domains	**References materials**
[a] Focus based on the North America	Gupta et al., 2017; Kannel et al., 2007; Oni and Fasakin, 2016; Panda et al., 2016; Sanchez et al., 2007; Yidana and Yidana, 2009; Banerjee and Srivastava, 2009; Ewaid and Abed, 2017; Gupta et al., 2016; Singh et al., 2018
[b] Application domain in USA	Bakan et al., 2010; Mladenovi´c-Ranisavljevi´c and Zerajic, 2018; Mojahedi and Attari, 2009; Ortega et al., 2016; Babaei Semiromiet al., 2011; Sanchez et al., 2007; Tomas et al., 2017; Zeinalzadeh and Rezaei, 2017
[c] Surface water, Soctland	Bordalo, 2001; Bordalo et al., 2006; Carvalho et al., 2011; Dadolahi-Sohrab et al., 2012; Ionus,, 2010
[d] This model developed based on the costeffective approaches	Dinius, 1987
[e] Evaluation of general water quality	References missing
[f] Model developed based on Spain	Pesce and Wunderlin, 2000; Koçer and Sevgili, 2014
[g] Oregon streams water, USA	Cude, 2001; Dunnette, 1979
Indices application Domains	**References materials**
[h] The Great lakes nearshore area, North America	Schierow and Chesters, 1988; Steinhurt and Somogniz, 1982
[i] The European community directives of specific uses purposes	House, 1980
[j] Surface water, New Zealand	Shah and Joshi, 2015; Smith, 1990
[k] The Vistula river basin, Poland	References missing
[l] Surface water bodies, Colombia state, USA	Zandbergen and Hall, 1998
[m] River water, southern Croatia	Nives, 1999, 2003
[n] Surface water, Canada	Saffranet al., 2001
[o] Keya river, Taiwan	Liou et al., 2004
[p] Streams water, USA	
[q] River water, Malaysia	Fulazzaky et al., 2010; Othman and Alaa Eldin, 2012; Amneera et al., 2013; Hasan et al., 2015; Naubi et al., 2016
[r] Surface water Vietnam	Pham et al., 2011

[s] The Potrero de los Funes river, Argentina	Almeida et al., 2012
[t] Java Sea, Indonesia	Sutadian et al., 2017

CONCLUSION

Water, one of the limited resources, has started to decrease in recent years due to global warming and water pollution. As a result, there is a risk of water scarcity in the near future. WQI are useful tools for the assessment of WQ and water management. WQI using various physico-chemical and biological parameters have emerged as a result of research and development conducted by different government agencies and experts around the world. WQIs are mathematical instruments or formulations that enable the aggregation and conversion of a dataset or parameters into a single value or dimensionless measure also known as composite indices, whose values usually range from 0 to 100. WQ rating according to different WQI methods is given in Table 11.

Table 11. *WQ Rating as per different WQI methods (Tyagi et al., 2013)*

WQI Value	Water Quality Rating
National Sanitation Foundation Water Quality Index (NSFWQI)	
91-100	Excellent WQ
71-90	Good WQ
51-70	Medium WQ
26-50	Bad WQ
0-25	Very bad WQ
Canadian Council of Ministers of the Environment Water Quality Index (CCME WQI)	
95-100	Excellent WQ
80-94	Good WQ
60-79	Fair WQ
45-59	Marginal WQ
0-44	Poor WQ
Oregon Water Quality Index (OWQI)	
90-100	Excellent WQ
85-89	Good WQ
80-84	Fair WQ
60-79	Poor WQ
0-59	Very poor WQ

References

Abbasi, S.A. 2002. Water Quality Indices, State of the art report, Scientific Contribution No. INCOH/SAR-25/2002, Published by-INCOH, National Institute of Hydrology, Roorkee,73p.

Abbasi, T. and Abbasi, S.A. 2012. Water Quality Indices; Elsevier: Kidlington Oxford, UK, pp.4-63.

Adelagun, R.O.A., Etim, E. E. and Godwin, O.E. 2021. Application of Water Quality Index for the Assessment of Water from Different Sources in Nigeria. Intechopen,1-19.

Akhtar, N., Ishak, M.I.S., Ahmad, M.I., Umar, K., Md Yusuff, M.S., Anees, M.T., Qadir, A. and Ali Almanasir, Y.K. 2021. Modification of the Water Quality Index (WQI) process for simple calculation using the Multi-Criteria Decision-Making (MCDM) Method: a review. Water, 13,90,1-34.

Akkoyunlu, M.E. 2012. Akiner Pollution Evaluation in Streams Using Water Quality Indices: A Case Study from Turkey's Sapanca Lake Basin, Ecol. Indic.18,501-511.

Al Yousif, M.A. and Chabuk, A. 2023. Assessment Water Quality Indices of Surface Water for Drinking and Irrigation Applications–A Comparison Review. Journal of Ecological Engineering, 24,5,40-55.

Almeida, C., Gonzalez, S.O., Mallea, M. and Gonzalez, P. 2012. A recreational water quality index using chemical, physical and microbiological parameters. Environ. Sci. Pollut. Res. 19,3400-3411.

Amneera, W., Najib, N.W., Z., Mohd Yusof, S.R. and Ragunathan, S. 2013. Water quality index of Perlis River, Malaysia. Int. J. Civ. Environ. Eng. 13,1-6.

Asian Water Development Outlook (AWDO), 2018. Strengthening Water Security in Asia and the Pacific. 2016. Available online: https://www.adb.org/publications/asian-water-development-outlook-2016 (accessed on 27 August 2018).

Babaei, Semiromi, F., Hassani, A.H., Torabian, A., Karbassi, A.R. and Hosseinzadeh Lotfi, F. 2011. Evolution of a new surface water quality index for Karoon catchment in Iran. Water Science and Technology, 64,12,2483-2491.

Bakan, G., Ozkoç, H.B., Tülek, S. and Cüce, H. 2010. Integrated Environmental Quality Assessment of the Ki{dotless}zi{dotless}li{dotless}rmak River and its Coastal Environment. Turkish J. Fish. Aquat. Sci. 10,453-462.

Balan, I.N,, Shivakumar, M. and Kumar, P.D.M. 2012. An assessment of ground water quality using water quality index in Chennai, Tamil Nadu, India. Chronicles Young Scient 3,2,146-150.

Banerjee, T. and Srivastava, R.K. 2009. Application of water quality index for assessment of surface water quality surrounding integrated industrial estate-Pantnagar. Water Sci. Technol. 60,8, 2041-2053.

Bhargava, D.S.1983a. Use of water quality index for river classification and zoning of Ganga River. Environ Pollut Ser B Chem. Phys. 6,51-67.

Bhargava, D.S. 1983b. A light-penetration model for the rivers Ganga and Yamuna. Int J Dev Technol 1,199-205.

Bharti, N. and Katyal, D. 2011.Water quality indices used for surface water vulnerability assessment. Int J Environ Sci. 2,154-173.

Bordalo, A. 2001. Water quality and uses of the Bangpakong River (Eastern Thailand). Water Res. 35,3635-3642.

Bordalo, A.A., Teixeira, R., Wiebe, W.J. 2006. A water quality index applied to an international shared river basin: The case of the Douro River. Environ. Manage. 38,910-920.

Brown, R.M., McClelland, N.I., Deininger, R.A. and Tozer, R.G. 1970. A water quality index-do we dare. Water Sew Work 117,339-343.

Brown, R.M., McClelland, N.I., Deininger, R.A. and O'Connor, M.F. 1972. A water quality index-Crashing the psychological barrier. In Indicators of Environmental Quality; Springer: Boston, MA, USA, 173-182.

Brown, R.M., McClelland, N.I., Deininger, R.A. and Landwehr, J.M. 1973. Validating the WQI. The Paper Presented at National Meeting of American Society of Civil Engineers on Water Resources Engineering. American Society of Civil Engineers, Washington, DC.

Brown, D. 2019. Oregon Water Quality Index: background, analysis and usage. State of Oregon Department of Environmental Quality, Laboratory and Environmental Assessment Program

Carvalho, L., Cortes, R. and Bordalo, A.A. 2011. Evaluation of the ecological status of an impaired watershed by using a multi-index approach. Environ. Monit. Assess. 174, 493-508.

CCME, 2001. Canadian Water Quality Guidelines for the Protection of Aquatic Life: CCME Water Quality Index 1,0 Technical Report, Canadian Environmental Quality Guidelines Canadian Council of Ministers of the Environment, 13p.

Chapman, D. 1996. Water Quality Assessments: A Guide to the Use of Biota, Sediments and Water. Environmental Monitoring. Chapman, D. (ed.), Second Edition. UNESCO, WHO, and UNEP. E&FN Spon, London UK.

Chaturvedi, M.K. and Bassin, J.K. 2010. Assessing the water quality index of water treatment plant and bore wells, in Delhi, India. Environ. Monit. Assess. 163,449-453.

Chidiac, S., El Najjar, P., Ouaini, N., El Rayess, Y. and El Azzi, D. 2023. A comprehensive review of water quality indices (WQIs): history, models, attempts and perspectives, Rev Environ Sci Biotechnol, 22,349-395.

Cude, C.G. 2001. Oregon water quality index: A tool for evaluating water quality management effectiveness, Journal of American Water Resources Association, 37,1,125-137.

Dadolahi-Sohrab, A., Arjomand, F., Fadaei-Nasab, M. 2012. Water quality index as a simple indicator of watersheds pollution in southwestern part of Iran. Water Environ. J. 26, 445-454.

Deininger, R.; Landwehr, J. A Water Quality Index for Public Water Supplies; Unpublished Report; School of Public Health, University of Michigan: Ann Arbor, MI, USA, 1971.

Dinius, S. 1972. Social accounting system for evaluating water resources. Water Resour Res. 8,1159-1177.

Dinius, S.H. 1987. Design of an Index of Water Quality. JAWRA J. Am. Water Resour. Assoc. 23,833-843.

Dunnette, D.A. 1979. A Geographically Variable Water Quality Index Used in Oregon. Water Pollution Control Federation, 51,1,53-61.

Ewaid, S.H. and Abed, S.A. 2017. Water quality index for Al-Gharraf River, southern Iraq. Egyptian Journal of Aquatic Research,43,2,117-122.

Fulazzaky, M.A., Seong, T.W. and Masirin, M.I.M. 2010. Assessment of water quality status for the selangor river in Malaysia. Water. Air. Soil Pollut. 205,63-77.

Gupta, S., Nayek, S. and Chakraborty, D. 2016. Hydrochemical evaluation of Rangit river, Sikkim, India: using Water Quality Index and multivariate statistics. Environ Earth Sci. 75,567,1-14

Gupta, N., Pandey, P. and Hussain, J. 2017. Effect of physicochemical and biological parameters on the quality of river water of Narmada, Madhya Pradesh. India. Water Sci. 31,11-23.

Hasan, H.H., Jamil, N.R. and Aini, N. 2015. Water Quality Index and Sediment Loading Analysis in Pelus River, Perak. Malaysia. Procedia Environ. Sci. 30,133-138.

Horton, R.K. 1965. An index-number system for rating water quality. J Water Pollut Con F. 37, 292-315.

House, M.A. 1980. A water quality index for rivers. Water Int. 5,16–21.

House, M. and Newsome, D. 1989. Water quality indices for the management of surface water quality. Water Sci Technol. 21,1137-1148.

https://consciouswater.ca/what-is-water-pollution/)

Icaga, Y. 2007. Fuzzy evaluation of water quality classification. Ecological Indicators,7,3,710-8.

Ionus¸, O. 2010. Water Quality Index-Assessment Method of the Motru River Water Quality (Oltenia, Romania). Geogr. Univ. DIN CRAIOVA Ser. Geogr. 13,74–83.

Johnson, D.L., Ambrose, S.H, Bassett, T.J, Bowen, M.L, Crummey, D.E, Isaacson, J.S, Johnson, D.N., Lamb, P., Saul, M. and Winter-Nelson, A.E. 1997. Meanings of Environmental Terms. Journal of Environmental Quality, 26, 581-589.

Kachroud, M., Trolard, F., Kefi, M., Jebari, S. and Bourrie, G. 2019. Water Quality Indices: Challenges and Application Limits in the Literature. Water, 11, 361,1-26.

Kannel, P.R., Lee, S., Lee, Y.S., Kanel, S.R. and Khan, S.P. 2007. Application of Water Quality Indices and Dissolved Oxygen as Indicators for River Water Classification and Urban Impact Assessment. Environ. Monit. Assess. 132, 93-110.

Kizar, F.M. 2018. A comparison between weighted arithmetic and Canadian methods for a drinking water quality index at selected locations in shatt al-kufa. IOP Conf Ser: Mater Sci Eng. 433,012026,1-14.

Koçer, M.A.T. and Sevgili, H. 2014. Parameters selection for water quality index in the assessment of the environmental impacts of land-based trout farms. Ecological Indicators, 36, 672-681.

Liou, S., M., Lo, S.-L. and Wang, S.H. 2004. A Generalized Water Quality Index for Taiwan. Environ. Monit. Assess. 96,35-52.

Lirika, K., Alma, I., Magdalena, C. and Dashnor, K. 2013. Use of diatom and macrophyte index to evaluate the water quality in Ohrid Lake, Journal of the Faculty of Engineering and Architecture of Gazi University,28,2, 393-400.

Lumb, A., Halliwell, D. and Sharma, T. 2002. Canadian water quality index to monitor the changes in water quality in the Mackenzie river-Great Bear". Proceedings of the 29th Annual Aquatic Toxicity Workshop, (Oct. 21-23), Whistler, B.C., Canada.

Lumb, A., Sharma, T.C. and Bibeault, J.F. 2011. A review of genesis and evolution of Water Quality Index (WQI) and some future directions. Water Qual Expo Health, 3,11-24.

Mladenovic-Ranisavljevic, I.I. and Zerajic, S.A. 2018. Comparison of different models of water quality index in the assessment of surface water quality. Int. J. Environ. Sci. Technol. 15, 665-674.

Mojahedi, S.A. and Attari, J. 2009. A Comparative Study of Water Quality Indices for Karun River. World Environ. Water Resour. Congr. 2009,1-9.

Naubi, I., Zardari, N.H., Shirazi, S.M., Ibrahim, N.F.B. and Baloo, L. 2016. Effectiveness of water quality index for monitoring Malaysian river water quality. Polish J. Environ. Stud. 25, 231-239.

Nives, S.G. 1999. Water quality evaluation by index in Dalmatia. Water Res. 33,3423-3440.

Nives, S.G. 2003. Comparison of Dalmatian Water Evaluation Indices. Water Environ. Res.75, 388-405.

Oni, O. and Fasakin, O. 2016. Open Access The Use of Water Quality Index Method to Determine the Potability of Surface Water and Groundwater in the Vicinity of a Municipal Solid Waste Dumpsite in Nigeria. Am. J. Eng. Res. (AJER). Am. J. Eng. Res. 96-101.

Ortega, D.J.P., Perez, D.A., Am´erico, J.H.P., De Carvalho, S.L. and Segovia, J.A. 2016. Development of index of resilience for surface water in watersheds. J. Urban Environ. Eng. 10, 72–82.

Othman, F. and Alaa Eldin, M.E. 2012. Assessment of the Klang River Quality Using the Water Quality Indices. Adv. Mater. Res. 599, 237-240.

Panda, P.K., Panda, R.B. and Dash, P.K. 2016. Assessment of Water Quality Index of River Salandi at Hadagada Dam and Its Down Stream upto Akhandalmani, Bhadrak, Odisha, India. American Journal of Water Resources, 4,2,44-53.

Paun, I., Cruceru, L., Chiriac, F.L., Niculescu, M., Vasile, G.G. and Marin, N.M. 2016. Water quality indices—methods for evaluating the quality of drinking water. In: Proceedings of the 19th INCD ECOIND International Symposium—SIMI 2016, "The Environment and the Industry", Bucharest, Romania, 13–14 October 2016,395-402.

Pesce, S.F., and Wunderlin, D.A. 2000. Use of water quality indices to verify the impact of Cordoba City (Argentina) on Suquia River. Water Research, 34, 2915-2926.

Pham, T.M.H., Sthiannopkao, S., Ba, D.T. and Kim, K.W. 2011. Development of Water Quality Indexes to Identify Pollutants in Vietnam's Surface Water. J. Environ. Eng.137, 273-283.

Prati, L., Pavanello, R. and Pesarin, F. 1971. Assessment of surface water quality by a single index of pollution. Water Res. 5,741-751.

Ramakrishnaiah, C., Sadashivaiah, C. and Ranganna, G. 2009. Assessment of water quality index for the groundwater in Tumkur Taluk, Karnataka State. Indian J. Chem. 6,523-530.

Rana, R. and Ganguly, R. 2020. Water quality indices: challenges and applications-an overview. Arabian Journal of Geosciences, 13, 1190,1-11.

Roozbahani, M.M. and Boldaji, M.N. 2013. Water quality assessment of Karoun river using WQI. Int Res J Appl Basic Sci. 5,628–632.

Saffran, K., Environment, A., Cash, K. and Canada, E.. 2001. Canadian Water Quality Guidelines for the Protection of Aquatic Life CCME Water Quality Index 1 . 0 User's Manual. Quality,1–5.

Samadi, M.T., Sadeghi,.S., Rahmani, A. and Saghi, M.H. 2015. Survey of water quality in Moradbeik river basis on WQI index by GIS. Environ Eng Manag, J. 2,7-11.

Sanchez, E., Colmenarejo, M.F., Vicente, J., Rubio, A., García, M.G., Travieso, L. and Borja, R. 2007. Use of the water quality index and dissolved oxygen deficit as simple indicators of watersheds pollution. Ecol. Indic. 7,315-328.

Sargaonkar, A. and Deshpande, V. 2003. Development of an overall index of pollution for surface water based on a general classification scheme in Indian context. Environ Monit Assess. 89,43-67.

Schierow, L.J. and Chesters, G. 1988. Evaluation of the great lakes nearshore index. Water Res. 22, 269–27

Scopus, 2022. Analyze search results Retrieved February 22, 2023, from https://www.scopus.com/term/analyzer.uri?sid=8eeff2944308f3417393fe6b0de5b7e1&origin=resultslist&src=s&s=TITLE-ABS-KEY%28water+quality+index%29&sort=cp-f&sdt=b&sot=b&sl=34&count=38419&analyzeResults=Analyze+results&txGid=68cf75652b70f07c51075648639736f3

Shah, K.A. and Joshi, G.S. 2015. Evaluation of water quality index for River Sabarmati, Gujrat. India App.l Water Sci.7,1349-1358.

Sharma, P., Meher, P.K., Kumar, A., Gautam, Y.P. and Mishra, K.P. 2014. Changes in water quality index of Ganges river at different locations in Allahabad. Sustain Water Qual. Ecol. 3-4,67-76.

Silvert, W. 2000. Fuzzy indices of environmental conditions. Ecol Model, 130,111-119.

Singh, P.K., Tiwari, A.K., Panigary, B.P. and Mahato, K. 2013. Water quality indices used for water resources vulnerability assessment using GIS technique: a review. Int J Earth Sci Eng. 6,1594–1600.

Singh, P.K., Panigrahy, B.P. and Verma, P. 2018. Evaluation of the Surface Water Quality Index of Jharia Coal Mining Region and Its Management of Surface Water Resources, in: Environmental Pollution. Springer Nature Singapore Pte Ltd. 429-437.

Smith, D.G. 1990. A better water quality indexing system for rivers and streams. Water Research, 24,10,1237-1244.

Steinhart, C.E., Schierow, L.J. and Sonzogni, W.C. 1982. An environmental quality index for the great lakes. JAWRA Journal of American Water Resources Association,18,6,1025-1031.

Suryawnshi, G. S., Kamble, S.S., Bhandare, A.N., Aniket, Tamboli, S.A., Sadikali, A.P., R. Chetan, R.P., Potdar, R.D. and Tamboli, A.E. 2018. Water Quality Index of Ground Water, International Journal of Engineering Science and Computing, 8, 4, 16636-16643.

Sutadian, A.D., Muttil, N., Yilmaz, A.G. and Perera, B.J.C. 2016. Development of river water quality indices-A review. Environ. Monit. Assess. 188,58,1-29.

Sutadian, A.D., Muttil, N., Yilmaz, A.G. and Perera, B.J.C. 2017. Using the Analytic Hierarchy Process to identify parameter weights for developing a water quality index. Ecol. Indic. 75, 220-233

Tirkey, P., Bhattacharya, T. and Chakraborty, S. 2015. Water quality indices-important tools for water quality assessment: A review. Int J. Adv. Chem.1,15-28.

Tiwari, T. and Mishra, M. 1985. A preliminary assignment of water quality index of major Indian rivers. Indian J. Environ Prot. 276-279.

Tomas, D., Curlin, M. and Mari´c, A.S. 2017. Assessing the surface water status in Pannonian ecoregion by the water quality index model. Ecol. Indic. 79,182-190.

Tyagi, S., Sharma, B., Singh, P. and Dobhal, R. 2013. Water quality assessment in terms of water quality index. American Journal of Water Resource, 1:34–38.

Uddin, M.G., Nash, S. and Olbert, A.I. 2021. A review of water quality index models and their use for assessing surface water quality. Ecol Indic 122,107218,1-21.

Vasistha, P. and Ganguly, R. 2020. Water quality assessment of natural lakes and its importance: An Overview. Materials Today: Proceedings, 32,544-552.

World Health Organization (WHO) 2011. Guidelines for Drinking Water Quality.

World Health Organization (WHO). 2017. Guidelines for Drinking-Water Quality. 4th Edition, 539p.

World Health Organization (WHO)-UN. 2010. World Health Organization and United Nations Children's Fund Joint Monitoring Programme (JMP) for Water Supply and Sanitation. Progress on Sanitation and Drinking Water, WHO, Geneva.

Yadav, A.K., Khan, P. and Sharma, S.K. 2010. Water quality index assessment of groundwater in Todaraisingh Tehsil of Rajasthan State, India-a greener approach. J. Chemother,7,428–432.

Yidana, S.M. and Yidana, A. 2009. Assessing water quality using water quality index and multivariate analysis. Environ. Earth Sci. 59, 1461-1473.

Yogendra, K. and Puttaiah, E.T. 2008. Determination of water quality index and suitability of an urban waterbody in Shimoga Town, Karnataka. Proceedings of Taal 2007: The 12th world lake conference, 342, 346.

Zandbergen, P.A. and Hall, K.J. 1998. Analysis of the British Columbia Water Quality Index for watershed managers: A case study of two small watersheds. Water Qual. Res. J. Canada.

Zeinalzadeh, K. and Rezaei, E. 2017. Determining spatial and temporal changes of surface water quality using principal component analysis. J. Hydrol. Reg. Stud. 13,1-10

Zhang, L. 2019. Big data, knowledge mapping for sustainable development: a water quality index case study. Emerg Sci J. 3,249-254.