The Handbook of Environmental Chemistry 59
Series Editors: Damià Barceló · Andrey G. Kostianoy

Nikos Skoulikidis
Elias Dimitriou
Ioannis Karaouzas *Editors*

The Rivers of Greece

Evolution, Current Status and Perspectives

Springer

The Handbook of Environmental Chemistry

Founded by Otto Hutzinger

Editors-in-Chief: Damià Barceló • Andrey G. Kostianoy

Volume 59

More information about this series at http://www.springer.com/series/698

The Rivers of Greece

Evolution, Current Status and Perspectives

Volume Editors: Nikos Skoulikidis · Elias Dimitriou ·
Ioannis Karaouzas

With contributions by

F. Botsou · N. Chrysoula · E. Dimitriou · A.N. Economou · D. Hela ·
N. Kamidis · I. Karaouzas · A. Koltsakidou · I. Konstantinou ·
P. Koundouri · D. Lambropoulou · L. Maria · I.D. Mariolakos ·
A. Mentzafou · A. Papadopoulos · D. Reppas · M. Scoullos ·
V. Skianis · N. Skoulikidis · M. Styllas · G. Sylaios ·
C. Theodoropoulos · L. Vardakas · S. Zogaris

Springer

Editors
Nikos Skoulikidis
Institute of Marine Biological Resources and Inland Waters
Hellenic Centre for Marine Research
Anavissos, Greece

Elias Dimitriou
Institute of Marine Biological Resources and Inland Waters
Hellenic Centre for Marine Research
Anavissos, Greece

Ioannis Karaouzas
Institute of Marine Biological Resources and Inland Waters
Hellenic Centre for Marine Research
Anavissos, Greece

ISSN 1867-979X ISSN 1616-864X (electronic)
The Handbook of Environmental Chemistry
ISBN 978-3-662-55367-1 ISBN 978-3-662-55369-5 (eBook)
https://doi.org/10.1007/978-3-662-55369-5

Library of Congress Control Number: 2017954950

Printed on acid-free paper

This Springer imprint is published by Springer Nature
The registered company is Springer-Verlag GmbH, DE
The registered company address is: Heidelberger Platz 3, 14197 Berlin, Germany

The Handbook of Environmental Chemistry Also Available Electronically

The Handbook of Environmental Chemistry is included in Springer's eBook package *Earth and Environmental Science*. If a library does not opt for the whole package, the book series may be bought on a subscription basis.

For all customers who have a standing order to the print version of *The Handbook of Environmental Chemistry*, we offer free access to the electronic volumes of the Series published in the current year via SpringerLink. If you do not have access, you can still view the table of contents of each volume and the abstract of each article on SpringerLink (www.springerlink.com/content/110354/).

You will find information about the

– Editorial Board
– Aims and Scope
– Instructions for Authors
– Sample Contribution

at springer.com (www.springer.com/series/698).

All figures submitted in color are published in full color in the electronic version on SpringerLink.

Aims and Scope

Since 1980, *The Handbook of Environmental Chemistry* has provided sound and solid knowledge about environmental topics from a chemical perspective. Presenting a wide spectrum of viewpoints and approaches, the series now covers topics such as local and global changes of natural environment and climate; anthropogenic impact on the environment; water, air and soil pollution; remediation and waste characterization; environmental contaminants; biogeochemistry; geo-ecology; chemical reactions and processes; chemical and biological transformations as well as physical transport of chemicals in the environment; or environmental modeling. A particular focus of the series lies on methodological advances in environmental analytical chemistry.

Series Preface

With remarkable vision, Prof. Otto Hutzinger initiated *The Handbook of Environmental Chemistry* in 1980 and became the founding Editor-in-Chief. At that time, environmental chemistry was an emerging field, aiming at a complete description of the Earth's environment, encompassing the physical, chemical, biological, and geological transformations of chemical substances occurring on a local as well as a global scale. Environmental chemistry was intended to provide an account of the impact of man's activities on the natural environment by describing observed changes.

While a considerable amount of knowledge has been accumulated over the last three decades, as reflected in the more than 70 volumes of *The Handbook of Environmental Chemistry*, there are still many scientific and policy challenges ahead due to the complexity and interdisciplinary nature of the field. The series will therefore continue to provide compilations of current knowledge. Contributions are written by leading experts with practical experience in their fields. *The Handbook of Environmental Chemistry* grows with the increases in our scientific understanding, and provides a valuable source not only for scientists but also for environmental managers and decision-makers. Today, the series covers a broad range of environmental topics from a chemical perspective, including methodological advances in environmental analytical chemistry.

In recent years, there has been a growing tendency to include subject matter of societal relevance in the broad view of environmental chemistry. Topics include life cycle analysis, environmental management, sustainable development, and socio-economic, legal and even political problems, among others. While these topics are of great importance for the development and acceptance of *The Handbook of Environmental Chemistry*, the publisher and Editors-in-Chief have decided to keep the handbook essentially a source of information on "hard sciences" with a particular emphasis on chemistry, but also covering biology, geology, hydrology and engineering as applied to environmental sciences.

The volumes of the series are written at an advanced level, addressing the needs of both researchers and graduate students, as well as of people outside the field of "pure" chemistry, including those in industry, business, government, research establishments, and public interest groups. It would be very satisfying to see these volumes used as a basis for graduate courses in environmental chemistry. With its high standards of scientific quality and clarity, *The Handbook of Environmental Chemistry* provides a solid basis from which scientists can share their knowledge on the different aspects of environmental problems, presenting a wide spectrum of viewpoints and approaches.

The Handbook of Environmental Chemistry is available both in print and online via www.springerlink.com/content/110354/. Articles are published online as soon as they have been approved for publication. Authors, Volume Editors and Editors-in-Chief are rewarded by the broad acceptance of *The Handbook of Environmental Chemistry* by the scientific community, from whom suggestions for new topics to the Editors-in-Chief are always very welcome.

Damià Barceló
Andrey G. Kostianoy
Editors-in-Chief

Preface

Greece is a small mountainous country with a remarkably varied relief, complex geological structure, a rich palette of microclimates, and diverse aquatic ecosystems hosting particularly rich biodiversity.

There is an erroneous perception that Greece is a dry country. This perception is derived from the fact that large areas of East and Southeastern Greece, which are popular tourist destinations, face water scarcity. In reality, the Greek Peninsula contributes over the double of river runoff in the European Mediterranean Sea (16%) compared to the surface area of the country (7%). The country is characterised by numerous, diverse, and highly fragmented small to medium-sized mountainous rivers and streams, running through steep narrow valleys. Large lowland areas that are diffused within prevailing rift valleys are drained by medium and large perennial rivers, which frequently form extensive flood and deltaic plains. Semi-arid landscapes are marked by intermittent to episodic streams. When considering this highly variable landscape, the uniqueness and diversity of aquatic flora and fauna is not surprising.

Water is according to Thales of Miletus (c. 624 – c. 546 BC) the originating principle of nature. Ancient Greeks defied rivers and created myths which conceal real physical-geological events. Since the second millennium BC, hydraulic and land reclamation works were conducted for water supply and protection against droughts and floods. Nowadays, to address the challenges of the unevenly spatial and temporal distribution of water resources, water managers diverted rivers and constructed numerous dams. Thus, the vast majority of medium and large rivers of the country are fragmented. The main pressures affecting running waters in Greece are hydromorphological modifications, agro-industrial wastewaters, agrochemicals, malfunctioning wastewater treatment plants, and, locally, mining. These pressures and particularly their combination with drought and water scarcity, triggered by gradual diminishing river flows, threaten lotic and riparian ecosystems.

Despite the vital importance of river ecosystems to the Greek civilization since ancient times, a comprehensive knowledge on their natural characteristics and diversity or the extent to which they have been exploited and degraded is limited.

This book volume is designed to provide a fundamental knowledge on the running waters of Greece covering topics related to potamology, either through means of review chapters or specific case studies. The topics covered include geomythology, biogeography, hydrology, hydrogeochemistry, hydrobiology, geomorphological, geological and biogeochemical processes, human pressures and ecological impacts, water management, both in the antiquity and today, and river restoration. This volume can be used as a basic or supplementary text in undergraduate and post-graduate courses or lectures in river ecology, river basin management, and conservation.

Acknowledgements

All chapters have been reviewed by the editors of this book volume and by a series of external reviewers which we are delighted to acknowledge here for their valuable help: Dr. Stefania Erba (IRSA-CNR, Italy), Dr. Nikolaos Nikolaidis (TUC), Dr. Niki Evelpidou (UOA, Greece), Dr. Angela Rouvalis (UOP, Greece), and Dr. Panagiotis Michalopoulos, Dr. Vasilis Kapsimalis, and Dr. Christos Anagnostou (HCMR, Greece). Finally, we would like to express our gratitude to Martha Papathanasiou (HCMR, Greece) for linguistic and grammar check of all contributions and to Dr. Andrea Schlitzberger (Springer DE) for her cooperation and support during the preparation of this volume.

Anavissos, Greece Nikos Skoulikidis
Anavissos, Greece Elias Dimitriou
Anavissos, Greece Ioannis Karaouzas

Contents

Ancient Greece and Water: Climatic Changes, Extreme Events, Water Management, and Rivers in Ancient Greece

Ilias D. Mariolakos

Abstract Climate change is not a phenomenon of our days, it is connected with the earth's history as indicated by both scientific evidence and ancient mythologies. Water, although essential for the survival of human kind, often triggers disasters and causes victims, mainly because of its unpredictable and uncontrollable nature. Especially in a country with a great history and a very old and long prehistory like Greece, its inhabitants have lived and experienced the climatic changes of the last 18,000 years and their dramatic geo-environmental impacts, such as sea-level rise, shoreline displacement, emergence and disappearance of springs, evolution and desiccation of lakes, and evolution and submergence of river deltas. All these disasters, coupled with landscape evolution, related mainly to the climatic–eustatic changes, are depicted in the Greek Mythology as the deification of the rivers, the struggle between heroes and springs, etc. A geomythological analysis of Greek myths has revealed that Greek Mythology is very old and is not just a figment of imagination of the resourceful Greeks, but it conceals real events. After the climatic stabilization (≈6,000 BP) and the cultural development of the Greek society, the main issue, besides water supply, was the protection against droughts and floods. This issue was addressed with the use of advanced geotechnical methods and hydraulic works.

Keywords Dardanus, Deucalion, Droughts, Floods (cataclysms), Geomythology, Heracles, Minyans

Contents

I.D. Mariolakos (✉)
Emeritus of Geology, National and Kapodistrian University of Athens, Athens, Greece
e-mail: mariolakos@geol.uoa.gr

N. Skoulikidis et al. (eds.), *The Rivers of Greece: Evolution, Current Status and Perspectives*, Hdb Env Chem (2018) 59: 3–30, DOI 10.1007/698_2017_474,

1 Introduction

Rivers, lakes, springs, and groundwater are very important physical–geological systems, as freshwater is necessary for the survival of human kind (water supply, irrigation, transportation, fishing, etc.). That is why human beings always preferred to live or settle near rivers, lakes, and springs all over the world. In Greece, two such lake settlements exist, already from the Neolithic Era; the first is located by the Kastoria Lake (N. Greece, W. Macedonia), the second one by the Viviis (Karla) Lake (E. Thessaly). Neolithic settlements have also been discovered along rivers, such as the Pineios and Aliakmon rivers.

Nevertheless, water sometimes became the cause of numerous problems to settlements, mainly due to its unpredictable nature related either to extreme rainfall events, or to the abrupt melting of snow or glaciers capping the high altitude mountainous areas. There are many examples of ancient cities that were flooded and finally disappeared, while the floods (cataclysms) narrations in the various myths mention even the extinction of entire nations. On the other hand, long-lasting droughts also shaped ancient civilizations.

Greek Mythology is the encrypted, most ancient history of the prehistoric inhabitants of the Aegean and Circum-Aegean region, as well as that of the neighboring areas. It describes effects of extreme climatic phenomena, which may serve to unravel prehistoric and historic climatic changes. The term geo-mythology has been introduced by Vitaliano [1]. Geo-mythology is a branch of the Geosciences, focusing on the interrelation between Geology and Mythology and particularly dealing with the physical–geological conditions during the mythological era [2]. The geomythological analysis of Heracles' works is proof that, in many cases, Greek Mythology depicts exactly the drastic changes of the geo-environmental regime due to major and minor climatic fluctuations since the beginning of the post-glacial period [18,000 years BP (Before Present)]. The deification of hydrological systems is connected directly to the climate, and mainly the climatic fluctuation of the last 18,000 years, i.e., since the last great global climatic change.

Since prehistory, ancient civilizations tried to control and tame water, while rivers were deified in Greek Mythology. After the stabilization of the global climate, during the period of the Holocene Climatic Optimum (6,000–5,000 BP) humans took advantage of the new favorable physical and geological conditions, in order to create the first permanent settlements in lowland areas, such as in closed basins, coastal areas, and river deltas. This is where they started to manage water by technical constructions designed to defend them against floodwaters. At the same time, humans started constructing systems for water supply and irrigation. Such technical works have been discovered in Egypt, Mesopotamia, and the Minoan Crete, dating as far back as the 3rd millennium B.C. Many of the anti-flooding technical works of prehistoric Greece differ from those of Mesopotamia and Egypt (Nile Delta) in that most of them had been constructed in closed hydrological systems, as, for example, in the basin of Kopais Lake. A great number of these works are mentioned in Greek Mythology, mainly in myths related to the God Poseidon or the hero Heracles.

Archeologists continue to discover hydraulic works dating back to the Minoan era used by the inhabitants, such as:

- Aqueducts, both underground and open, for the transportation of water collected at springs or wells, in order to supply cities.
- Dams for collecting water (reservoirs) and/or aiming at anti-flood protection.
- Canals of several kilometers' length, for the drainage of lakes.
- Construction of tunnels or usage of caves for the drainage of floodwaters.
- The combined usage of a river, by constructing troughs filled with river water around settlements both for anti-flood protection and for defense purposes.
- Torrent regulation works.
- Land reclamation works at all river deltas.
- Sewage works, and much more.

It is noteworthy that even from the sixth century B.C., there was a special legislation regulating and managing water resources in the city of Athens. This was the work of the great legislator Solon, while the most ancient ceremony related to water is mentioned in Arcadia, dating back to the Mycenaean times (1600–1100 B.C.).

This chapter describes climate changes that took place after the last glacial maximum. It emphasizes extreme climatic phenomena depicted in the Greek Mythology and attempts a geomythological, physical, and geological dating of the oldest flood mentioned. It presents the hydraulic works during historic and prehistoric times using historical evidence and through the interpretation of related myths. It finally argues on the reasons of river deification in ancient times.

2 Climate and Climate Change

The accepted climate changes, up to some decades ago, were those of the quaternary (i.e., the last 2 million years) that include four glacial and four interglacial major periods. Since the time of the classical period of ancient Greece, those climate changes were well known. Plato (427–347 B.C.) was the first who mentioned that climatic changes exhibit a periodicity. According to *Milankovitch theory* [3], the reasons for the climate periodicity are mainly astronomical, related to the rotation of the Earth around the sun and around its own axis. These reasons are the following:

- *Eccentricity of the Earth's orbit*, with a period of ≈100,000 years
- *Obliquity of the Earth's axis*, with a period of ≈40,000 years
- *Precession of the Earth's axis* (⇒equinoxes) with a period of ≈23,000 years.

There are many methods for the assessment of the palaeoclimatic conditions such as palaeo-temperature and palaeo-humidity. Among them, the most common are: (1) the paragenesis of the clay minerals in the palaeosoils, (2) pollen analysis, and (3) the ratio of $^{18}O/^{16}O$, based on measurements in shells of marine organisms or in trapped air bubbles in ancient glaciers. Based on the previous methods, a periodic change of temperature and humidity provides indirect information about precipitation.

Apart from the aforementioned major glacial/interglacial periods, there are more periods, cycles of shorter duration, which are controlled from other factors as well. In Fig. 1, the periodic changes of the mean global temperature are depicted for the time period between 18,000 BP and the present.

The main reasons and consequences of climatic changes are presented in Fig. 2. From the factors mentioned, the ones that, directly and/or indirectly, influenced Greek Mythology and Greek culture are those relating to rivers, droughts, floods (cataclysms), and the sea-level changes, the so-called *climate–eustatic movements*. All these have drastically influenced the life of Homo sapiens for the time period between approx. 18,000 BP and 6,000 years BP, that is, until the "Climatic Optimum of the Holocene." Nevertheless, much later, and even nowadays, climate remains the main factor controlling the cultural, social, and economic development of a region.

3 Dry and Wet Periods in Greek Mythology

3.1 Droughts

Because of the fluctuation of precipitation (humidity), as a result of a small-scale climatic periodicity, droughts are often mentioned in relation to cataclysms in various myths.

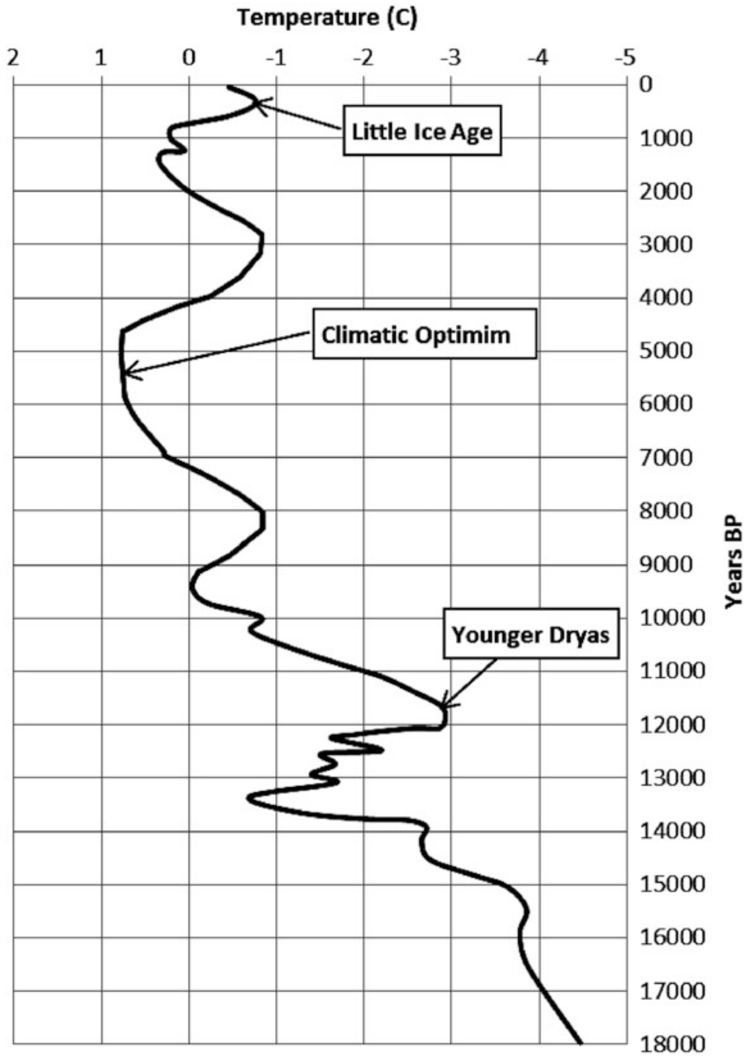


Fig. 1 Mean global temperature fluctuation for the last 18,000 years (based on [4])

From Greek Mythology, the best-known droughts are the following:

1. The Inachus Drought, which was, most probably, the oldest one. After the Greek writer Apollodorus [26], it was manifested when Inachus was king of Argos during the Mycenaean period, but it dates long before the foundation of Mycenae city.

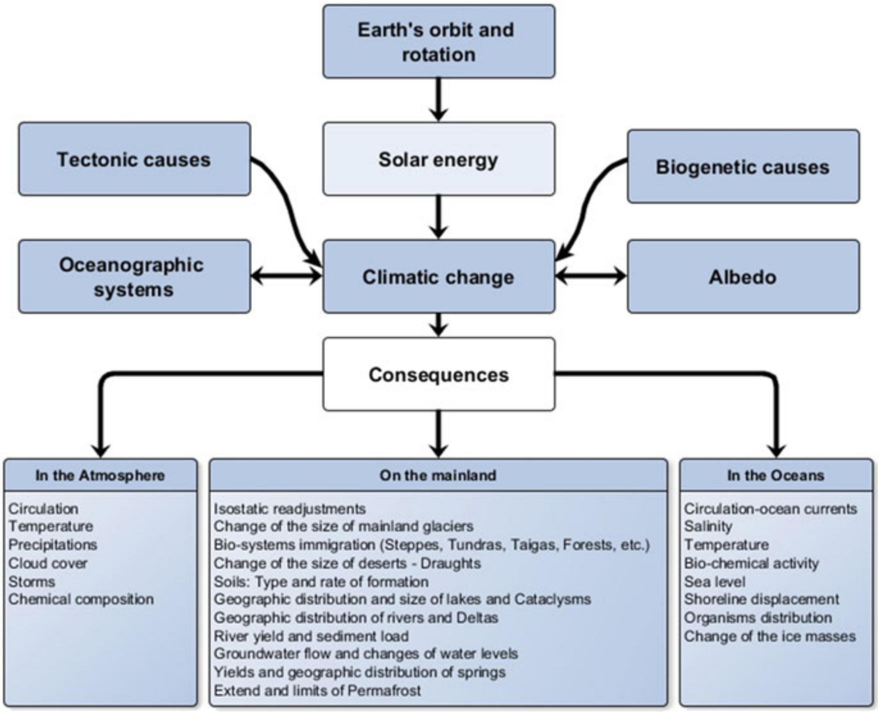


Fig. 2 The main reasons and consequences of climatic changes

2. The Danaus Drought, which dates after the return of Danaus from Egypt. Danaus, the king of Argos, left Egypt around the same time of Moses's Exodus, i.e., around the end of the fourteenth century B.C.
3. The Aeakeios Drought (know from the Greek Mythology), which occurred when Aiakos was the king of the prehistoric Aegina Island in the Saronic Gulf.
4. The Agora Drought, dated in eighth century B.C. This drought was the reason that many shrines were built up, dedicated to Omvrios (rainy) Zeus, at different places in eastern Greece, e.g., at Aegina island, at Acropolis, at the eastern slopes of Hymettus mountain (east of Athens) and elsewhere. During this drought period, the Athenians addressed a prayer to Zeus: "Υσον Υσον ω φίλε Ζευ, κατά της Αρούρης των Αθηναίων και των πεδίων...", i.e., "Rain, rain dear Zeus on the Earth of the Athenians and the plains..."

3.2 *Flood Periods: Cataclysms*

The causes of a cataclysm are various. In Greece, the most important and most frequent are the catastrophic cataclysmic rainfalls lasting for relatively long time periods. A cataclysm of an area potentially also relates to other phenomena, such as the manifestation of a tsunami, the rapid glacier meltdown, and the subsidence of a coastal region due to an earthquake. Noah's flood, mentioned in the Old Testament, is the most widely known cataclysm. Many people are left with the impression that Noah's flood was the only one in human history. This is incorrect, however, since cataclysms are mentioned in the mythologies of many nations all around the world. During the prehistoric era of Greece, at least three great floods were recorded, i.e., the Deucalion flood, the Ogyges flood, and the Dardanus flood.

3.2.1 The Flood of Deucalion

Deucalion was the son of Prometheus and Pandora or Clymene, daughter of the Titan Atlas. Since the offspring of the Titans were disrespectful criminals, just like their forefathers, Zeus decided to eliminate their breed, a decision that was hailed also by the other Gods.

When Prometheus learned about the Gods' decision, he advised his son Deucalion and Deucalion's wife, on how to save themselves, since they were devout and just. Thus, Deucalion constructed a vessel and, along with his wife Pyrrha, daughter of Epimetheus (Prometheus' brother), and others, locked themselves in it. Zeus sent so much rain that most places in Greece were flooded and all the people and animals perished except a few who found refuge on high mountains. It is said that during this time, the mountains of Thessaly, Olympus, and Ossa (Fig. 5) were separated (Apollodorus, A, 7) and the Tempi valley was formed by the floodwaters that overflooded the Pinios River. Finally, after 9 days of suffering, Deucalion ended up on Parnassus Mt., at a small plateau above Delphi, known nowadays as "Livadi."

According to the Parian Chronicle (dated around 264–263 B.C.), Deucalion's flood must have taken place at 1529 B.C. If this dating is correct, then the flood must have occurred during the transition between Meso and Late Helladic period.

3.2.2 Ogygis Flood

Ogygis or Ogygos was the name of the first indigenous king of Boeotia and Attica and their most ancient inhabitants. In Attica, he was considered the son of Gaia, while others believed that he was the son of Poseidon or Boeotos because, at the same time, he also reigned in Thebes. It is said that, during Ogygis' reign over Attica, the first flooding of Greece took place, which devastated mainly Attica and Boeotia. Regarding the dating of this flood there are several theories that place it between 1796 and 2136 B.C., without excluding the possibility that it occurred much earlier, i.e., between 12,500 and 14,500 BP [5].

3.2.3 Dardanus Flood: A Geomythological, Physical, and Geological Dating Approach

In fact, there are two myth versions related to two floods with the name Dardanus flood. The first is the one which took place in Arcadia (Central Peloponnese), whereas the second one on the island of Samothrace. In both cases the protagonist is Dardanus.

1st Version: Dardanus Flood of Arcadia

According to the first version, which is supported by various authors and among these Strabo [28] (64 B.C.–ca. 23 A.D.), Dardanus was born in Arcadia. It is said that Electra, the daughter of the titan Atlas, who, at that time, was the king of a part of Arcadia, gave birth to two sons of Zeus, named Iasion and Dardanus, and a daughter, Armonia (Harmony).

It is mentioned that during that time a cataclysm took place that had as a consequence to inundate the plains of Arcadia, where a great number of hydrologically closed basins exist, that is great karstic surface sinking, known as poljes. Because of the flood, all the residents had to move uphill to the mountains. However, as the mountains could not cover the dietary needs of the people, only the son of Dardanus, Dimas, with some of the residents remained in Arcadia, whereas Dardanus and Iasion, with the rest of the residents left and moved to the island of Samothrace.

2nd Version: SAMOTHRACE: Dardanus Flood and the First Flooding of Black Sea

Beyond the version of the flooding that occurred in Arcadia during the Dardanus epoch, Diodorus Siculus (90–30 B.C.) mentions another flooding event, in which Dardanus was involved. Diodorus Siculus describes this flood in such detail, that there is no doubt about the event and the affected region. The following text comes from the most important work of Diodorus Siculus, known as "Historical Library [27]."

> ...And the Samothrace people are telling a story according which in their area a great cataclysm occurred earlier than the cataclysms in other nations. This flood occurred because initially the Cyanean Rocks opened where the Symplegades were estimated to be located (the mouth towards Bosporus strait) (Fig. 3). The Hellespontus (Dardanelles strait) followed. This occurred because the Euxinean Pontus (Black Sea) that till then was a lake since the communication with the sea has ceased, it flooded by river waters that were collected in it and as a result its level rose so much, that overflooded the Bosporus strait, and flooded Propontis (Marmara Sea). And in its turn, because it overflooded the Hellespontus reached the Aegean Sea and as a consequence a large coastal region was flooded, not only in Asia but Samothrace as well. For this reason, the following years, fishermen were finding stone capitals in their fishnets since cities were also flooded. At the same time residents that

witnessed the flood in order to save their lives, run uphill to the higher altitude locations of the island. As the sea level continued to rise the people prayed to the gods and were saved while in memory of their salvation they delimited with rocks the island and founded altars that made sacrifices that continue in our days...

From this text, it is obvious:

- That this flood is the most ancient one ("... a great cataclysm occurred earlier than the cataclysms in other nations...").
- The water came from the *Black Sea*.
- This happened because the outlet of the *Cyanean Rocks* (=outlet to Bosporus) *opened, and*
- Because the sea level of the *Euxinenan Pontus (Black Sea) rose*. This caused the overflow and the flooding of Bosporus. The water then overflowed the *Propontis (Marmara Sea)* then the *Dardanelles* and finally the water overflowed the Aegean Sea, part of which was a land (Fig. 3).
- The overflow of the Black Sea resulted "... by the river water which flow into it..."

From this amazing, revealing, and scientifically interesting story of Diodorus Siculus [27], we can extract a geomythological interpretation of what is mentioned. Thus, it appears that: (1) During the prehistoric times, and for a certain time period, the communication between the Aegean and the Black Sea must have ceased (Table 1 and Fig. 3). This cessation must have happened when the sea level in the Black Sea and the Aegean Sea was lower compared to the highest level of the bottom of Bosporus and Dardanelles. (2) The present-day Bosporus and Dardanelles straits were valleys, obviously drained by a river.

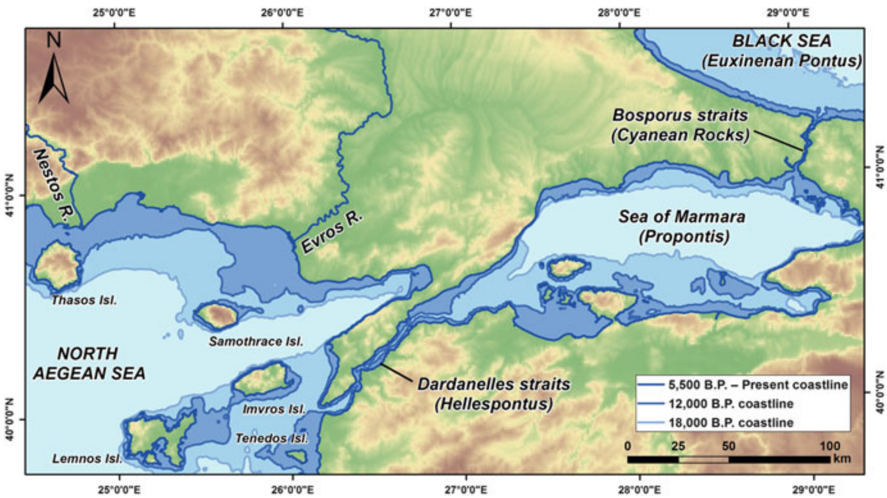


Fig. 3 Palaeogeographic evolution of the North-Eastern Aegean Sea during the last 18,000 years, caused by the sea-level changes (18,000 BP coastline: *cyan area*. 12,000 BP coastline: *blue area*. 5,500 BP – Present coastline)

Table 1 The approximate global sea level during the last 18,000 years, due to the glacio-eustatic movements [6]

Approximate global sea-level		
7,500–6,800 BP	≈	−10 m
9,000–6,800 BP	≈	−15 m
9,400–7,300 BP	≈	−20 m
9,800–7,700 BP	≈	−25 m
10,200–8,200 BP	≈	−30 m
11,300–8,500 BP	≈	−35 m
10,000–9,000 BP	≈	−40 m
11,400–10,000 BP	≈	−45 m
12,200–11,400 BP	≈	−49 m
13,000–12,200 BP	≈	−52 m
13,500–13,000 BP	≈	−57 m
13,800–13,500 BP	≈	−61 m
14,000–13.800 BP	≈	−67 m
14,700–14,000 BP	≈	−73 m
15,000–14,700 BP	≈	−94 m
16,000–15,000 BP	≈	−100 m
18,000–16,000 BP	≈	−115 m
18,000 BP	≈	−125/−150 m

Physical and Geological Dating of the Samothrace Flood

One of the most important questions rising from the study of Diodorus' text is whether it is possible to verify this story. In other words, is this cataclysm actually the oldest of all mentioned in world mythologies?

Despite the purpose of several projects which applied the state-of-the-art geological techniques of sediment analysis, seismic profiles and dating of the samples which were taken from drill cores in different points and different depths from the Black Sea [7–11], the occurrence, timing and direction of possible flood events from the Aegean to the Black Sea and vice versa since the last glacial maximum is still debatable [8, 12–15]. However, scientists agree that during the oldest flood melt water moved from the Black Sea towards the Aegean Sea [12, 15].

In order to date the Samothrace flood, a method named "physical-geological dating method" [16] has been used. To apply this method, we need to know:

1. The climatic changes of the last 18,000 years (Fig. 1).
2. The way the sea level changed in the Aegean and the Mediterranean Sea during the last 18,000 years (Figs. 3 and 4, and Table 1).
3. The changes of the water quantities and those of the landscape, such as:
   a. The hydrographic systems (rivers) that flow into the Black Sea.
   b. The sea bottom relief of Bosprorus straits, Sea of Marmara (Propontis), and that of the Dardanelles straits (Hellespontus).
   c. The coastal changes in the wider area around Samothrace Island and the north-eastern Aegean Sea, due to the climatic–eustatic changes.

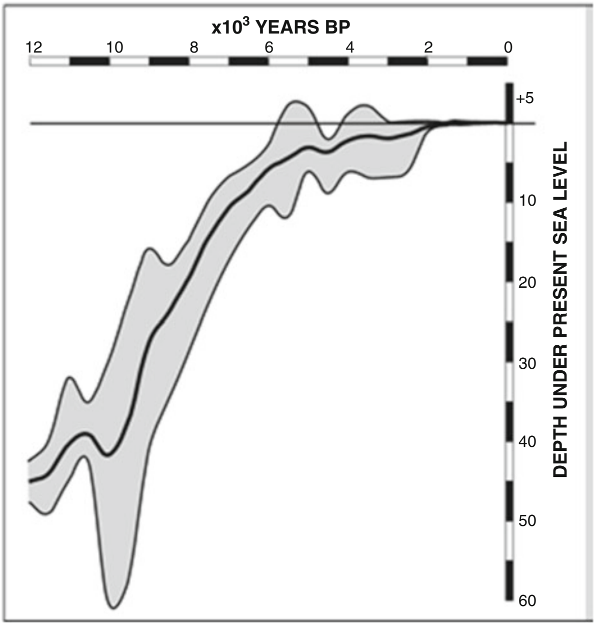


Fig. 4 The sea-level rise curve for the last 12,000 years [17]

The reasoning for the dating of the Samothrace flood, taking into account all the aforementioned, is as follows: Since the present sea level and consequently the hydraulic regime between the Black Sea and the Aegean Sea was finalized during the Climatic Optimum of the Holocene, which happened about 6,000 years ago, then the flood of the Aegean by the freshwater of the Black Sea must have taken place before this period. This is the necessary preposition, in order for the freshwaters to arrive to the Aegean Sea from the Black Sea, as the sea level of the Aegean should have been lower, compared to that of the Black Sea. Consequently, the first partial conclusion, and at the same time the first dating approach, is that the flood mentioned by Diodorus must have happened earlier than 6,000 BP. But how much earlier?

As mentioned above, in order for the water to flow from the Black Sea to the Aegean Sea, it is necessary not only for the Aegean Sea level to be lower than the present-day one, but also for it to be lower than the highest area of the Bosporus bottom, i.e., the area that formed a morphological swell, i.e., a "natural barrier," which blocked the hydraulic communication between the Black Sea and the Sea of Marmara at first, and then between the latter and the Dardanelles straits. The present-day minimum depth of the Bosporus bottom at the location of this natural barrier is about 35 m. Consequently, if we accept that the morphological relief of the Bosporus seafloor has not changed dramatically during the last ca. 10,000 years, in order for the water to flow from the Black Sea into Bosporus and then to the Marmara Sea, the Black Sea level should have been slightly higher compared to the above-mentioned depth (35 m) and, at the same time, the Aegean Sea level should have been lower than the present-day one (lower than 35–40 m). This is a necessary condition since, if the global sea level was slightly higher than 35 m, then a hydraulic regime similar to the present-day would have been established, as the water from the Aegean Sea could flow to the Black Sea. Based on the data of the curves in Fig. 4 and Table 1, it can be proposed that the Aegean Sea level was at this point at about 11,000 BP. But this seems difficult to have happened, since the Earth had just come out of the Younger Dryas (12,500–11,400 BP) cool stadium, and as mentioned previously, the sea level at the Black Sea should have been lower than the Bosporus bottom ridge.

If we further accept that: (1) the Mediterranean sea level follows the changes of the global sea level, (2) the results from the studies carried out by Demek and Kukla [17], which have shown that around 8,000 BP, a dry climate dominated central Europe and consequently the yield of the rivers flowing into the Black Sea should have been significantly less than those of the previous periods, (3) the confirmation that, during the time period between the Younger Dryas period till 7,600 BP, the communication between the Black Sea and Aegean had ceased, (4) the highest sea-level at the Black and Caspian Seas was observed during the period before the cool period (stadial) of Younger Dryas, it emerges that the Samothrace flood, as described by Diodorus, and more specifically its first stage, should have happened earlier than 12,500 BP. But how much earlier?

Based on the temperature curve (Fig. 1), the rise of the sea level in the Black Sea should have started right after 18,000 BP, as soon as the increase of the global temperature and the melting of the continental glaciers. It is estimated that the sea level of the Black Sea could not have reached the highest point before 14,500 B.P. So, the Samothrace flood must have happened somewhere between 14,500 and 12,500 BP. This is in agreement with other scientific evidence [5, 15, 18, 19]. Consequently, the description of Diodorus Siculus for the approximate time period of this event seems to be correct. This means that the flood is, by many thousand years, older than the most popular cataclysms, such as that of Noah and Deucalion and many others.

When exactly the water flow from the Black Sea to the Aegean ceased is not possible to determine, but it is certain that at the beginning of the Younger Dryas glacial stadial, the hydraulic communication should have ceased, because the sea level in the Euxenean Pontus dropped so much, that the overflow over the "submarine barrier" of the Bosporus bottom was not possible.

If the findings of the physical–geological research that has been carried out till our days will not be revised from future studies, then these results are fascinating and change many views related to the ancient Greek civilization and prove that Greek Mythology is not a nice fairytale, a fiction of imagination of the ancient Greeks, but it represents the very old history of the inhabitants of this land which, many thousands of years later, was named Hellas (Greece).

4 Water Management in Ancient Greece

4.1 The Prehistoric Hydraulic Works

During the second millennium B.C., in the Hellenic region, many hydraulic and land reclamation works were carried out, such as drainage and flood prevention works constructed mainly during the Mycenaean period (1600–1180 B.C.). However, it is possible that many of these works may potentially date earlier than the Mycenaean period. The most famous works were constructed by the Minyans, mainly in the regions of Kopais, Aitoloakarnania, Argolis, Thessaly, and Arcadia. Figure 5 presents a map of Greece showing the locations mentioned in the text.

4.1.1 Hydraulic Works of the Minyans: The First Hydraulic Civilization in Europe

The Drainage Works in Kopais Basin (Boeotia)

The Minyans were a Protohellenic or Pelasgian tribe from Thessaly, which came to Boeotia and started to drain the ancient lake of Kopais, around sixteenth century B.C., whereas other archaeologists believe that all these works started long before the twentieth century B.C.

The Kopais basin represents a seismically active, neotectonic graben, formed by composite and continuous geological processes such as active tectonics (faults) and erosion of the carbonate rocks (karstification), which explains the large caves at its margins. It is a rather large polje. Two main rivers, amongst many other minor streams, meet in Kopais basin: the Boeotian Kifissos and the Melas Rivers. Today, the mean annual discharge of the Boeotian Kifissos River is estimated to be 200 million m^3. The Melas River carries the water of the karstic springs of Hariton (Orchomenos) and has a mean annual discharge of 80 million m^3. All this water quantity was discharged in the Kopais basin, which was transformed to a lake since it didn't have surface drainage and comprised a hydrologically closed system. Nowadays, this drainage is taking place through many minor draining troughs and a major one that transports water to Yliki Lake.

After Knauss [20, 21], "…the technical installations constructed and operated by the Minyans in the MH (2050 – 1550 B.C.) and LHIII period (≈ 1300 B.C.) in order to reclaim land from lake Kopais must be characterised as extraordinary and ingenious, fully justifying the claim for the "first hydraulic civilization in Europe". In fact, to drain the lake of Kopais, the Minyans started to gradually construct several flood prevention embankments and soil dams around the sixteenth century B.C. At the northern margin of the basin, they constructed a draining canal (Fig. 6) with a total length of 27 km. This canal, which was initiating from the city of Orchomenos, gathered the waters from Boeotian Kifissos and Melas rivers. The location of the diversion of the Boeotian Kifissos River towards Melas River must

Fig. 5 Map of Greece with locations of prehistoric hydraulic works and myths

have been in Orchomenos area. They had constructed a 3–4 m high earth embankment. It was 35 m wide at the base and around 30 m at the top, with a 100% impermeable core made of clay material. The core was reinforced by a less permeable casing, whereas the whole earth construction was covered by rock blocks, to avoid the erosion by water flow. They had also sealed the bottom of the canal. The canal water was transported to the northeastern margin of the basin and from there, through the large cave at Neo Kokkino (Fig. 7), known as "Heracles cave," it drained into the North Euboic Gulf. The most important feature, from a technical point of view, is that the water level within the canal was 1.5–2 m higher in relation to the bottom of the drained lake in which the Minyans began to

Fig. 6 The canal constructed by the Minyans during the Mycenaean period

Fig. 7 The entrance of the "Cave of Heracles" at Neo Kokkino

construct their towns. The canal was used for flood prevention, irrigation, and transportation [20, 21].

According to the Greek Mythology, Heracles from Thebes (Thivae) demolished a rock from the ceiling of the great cave-sinkhole of Neo Kokkino, a city next to Kopais basin, in order to destroy the anti-flooding works of Minyans, resulting to the sealing of the entrance and the destruction of the embankments due to overflowing. In this way, the Kopais basin turned back to a lake.

The Drainage Works in Arcadic Orchomenos

Three physical–geographical terrain types exist in Arcadia (Central Peloponnesus), i.e., the mountainous region, the flat land, and the coastal area. The flat land represents the known Arcadic plateau, which is composed of various drainage basins, the largest of which is the Tripolis plateau. Other smaller, similar basins are Stymfalia and Feneos. From a geological point of view, the Arcadic plateau is made of mainly intensely karstified carbonates. In many cases, the water is drained through sinkholes and from there to the karstic springs of Lerni, Kroi, Kiveri, and others, mainly near the coasts of Argolis. Each spring has its own history and myths. The basins of the Arcadic plateau are closed hydrological systems. As a result, many regions are being flooded for shorter or longer time periods. For the people to utilize this land, they had to find ways and techniques, not only to drain these areas in order to cultivate them, but also at the same time to use the water for irrigation. This is the reason why the prehistoric inhabitants of Arcadia constructed dams in every basin.

The prehistoric settlement of Arcadic Orchomenos was founded at the foot of the Acropolis, while during historic times the city was moved on a low mountain, where the most important monuments (Agora, Theater, etc.) are also situated. At the plain area, artificial constructions for drainage and for the rearrangement of river-beds during prehistoric and later times have been identified (Fig. 8).

Fig. 8 The prehistoric hydraulic works of the Minyans, on the slopes of the artificial trench, which may constitute watermill installations. In the background is the big Kandela polje – the Lower Field of Pausanias – where a dam and a reservoir have also been constructed. The road in the background has been constructed on the prehistoric earth dam

The Hydraulic Knowledge of the Minyans

As it is concluded from the study of the remains of hydraulic works, 4,000–3,500 years ago, the Minyans must have possessed good scientific and technical-geological knowledge. They had the knowledge of modern hydraulic engineers, hydrogeologists, etc., otherwise it would not have been possible for the embankment to be maintained until today.

They must have been familiar with:

- The behavior of the karstic formations (voids, caves, etc.)
- The flooding charges of rivers and overflowing mitigation.
- The rocks' and soils' physical and mechanical properties (permeability, cohesion, etc).
- The topographical methods, to estimate the gradient and the velocity of water, for the flow to be linear and not turbulent.
- The calculation of the dip, the slope stability, as well as the methods of erosion protection.
- The stratification and compaction of the soil.
- Knowledge of management and organization of large worksites.

4.1.2 Water Management in the Minoan Era

The urbanization in the Aegean and the Peri-Aegean area is very old, while freshwater resources are scarce in this area. This was the main reason to develop water resources management technologies, such as wells, groundwater hydrology, aqueducts, and cisterns [22, 23]. The best examples are the public works in Crete (Fig. 9), where the urbanization started as early as the Bronze Age (ca. 3500–2000 B.C.). Apparently, the knowledge of the hydraulic engineers of this very old time must have been very high. It is remarkable that the Minoans knew the use of siphon and the principle of the communicating vessels. The remains of their water installations are proof of the quality of their work since many closed pipes of that time were so flawless, that they are still in operation.

4.2 Hydraulic Works in the City of Athens

The city of Athens has a long history, whereas the urbanization had already started during the prehistoric times. To cover the freshwater needs of the city, the Athenians constructed many wells. Nevertheless, as the water demand was continuously increasing, they started to construct aqueducts, to transport freshwater from springs, located far from the ancient city (Fig. 10). The following are examples of some of these.

(a)

(b)

Fig. 9 Well (**a**) and cistern (**b**) from Zakros Minoan palace (eastern Crete)

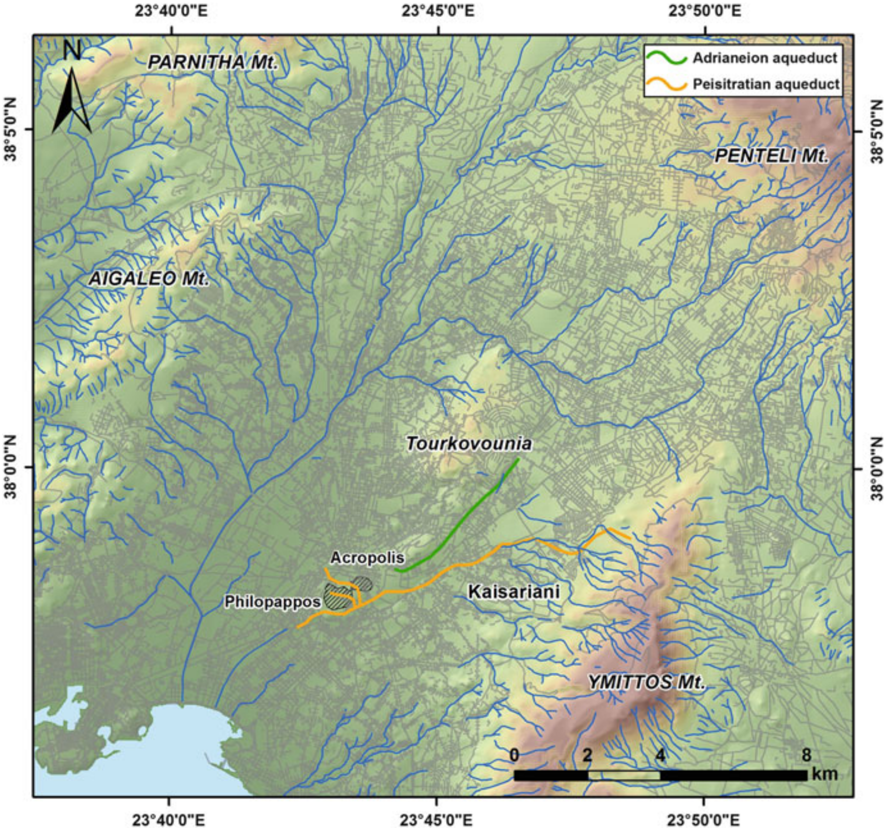


Fig. 10 Topography map of the Athens area, showing the trace of both the Peisitratian (ΠΕΙΣΙΣΤΡΑΤΕΙΟ) (*orange line*) and Adrianeion (ΑΔΡΕΙΑΝΕΙΟ) (*green line*) Aqueducts

4.2.1 The Pelasgian Aqueduct

It is the most ancient one. It must have been constructed long before the times of Heracles and Theseus, i.e., before the thirteenth century B.C. Its route began probably from the area of present-day Kaisariani and it was ending in the pass between the hills of Acropolis and Philopappos. The water would have come from the springs located on the NW flanks of Hymettus, the mountain to the east of Athens plateau.

4.2.2 The Aqueduct of Theseus

Theseus, who is considered the first king of Athens, lived in the beginning of the thirteenth century B.C., i.e., during the Mycenaean Times. He coexisted with

Heracles, but he was a bit younger. The water in Theseus aqueduct came from the springs located on the west side of Parnitha Mt.

4.2.3 The Aqueduct of Peisistratus

Based on the laws of *Solon* and the growing needs of the city of Athens due to the increase of population and living standards, Peisistratus (approx. 600–527 B.C.), a ruler – tyrant of Athens, constructed a great aqueduct between 540 and 530 B.C. The Peisitratian Aqueduct is still functional, more than 2,500 years later. The water originates from the springs of Hymettus Mt., located east of the Pelasgian aqueduct. Later, a great number of smaller aqueducts were constructed, to cover the needs of various parts of the growing city.

4.2.4 The Aqueduct of Adrianus (Adrianian Aqueduct)

In 117 A.D., Adrianus was crowned emperor of Rome. He is considered a benefactor of Greeks because of the many public works he had constructed in Athens, in Arcadia, in Corinth, and elsewhere. One of the most important ones was an aqueduct, named after him, the "Adrianian Aqueduct." Its construction begun in 125 A.D. and finished in 140 A.D. Adrianus never saw it completed, since he died in 138 A.D. The "Adrianian Aqueduct" supplied Athens with water for about 1,800 years. Its water came from springs located between the mountains of Parnitha and Penteli.

4.3 Solon's Water Supply Laws

Because of the lack of precipitation in Eastern Greece (where Athens is located) and the Cycladic islands (400–600 mm/year), besides the constructed technical works, laws were voted in order to protect and manage water resources. They were the laws of Solon (640–560 B.C.) and those of *Plato* (428–347 B.C.). Solon's "water supply" laws regulated the citizens' rights on water well use and the distance between water wells and their depth. According to these laws, a special service was established, as well as a special "water law enforcement" body comprising the following institutions:

1. The "Hydrognomon," that is the "water inspector,"
2. The "Crenophylax," the "fountains guard,"
3. The "Commissary of the fountains," who was elected, and was responsible for the city's policies on water supply. It is worth mentioning that Themistocles was elected in this honorary position before the naval battle of Salamis.

4. The "Agronomist," who was responsible for the maintenance and the irrigation of the fields, as well as for the safe keeping of the forests, and
5. The "Hydronomeus" who was responsible for the water allocation.

5 Heracles and Water

Heracles, the son of Zeus and Alcmene, the wife of Amphitryon, who was the King of Thebes, is perhaps the most famous mythical hero worldwide. In modern times, Heracles gives the impression of a heavily built man with muscles, a tough man dealing mainly with wild mythical creatures. This wrong impression of the hero is directly linked to the erroneous view that the Greek Mythology is a figment of imagination of the ancient Greeks.

Heracles is known mainly for his twelve labors. Beyond these labors, Heracles was also involved in other "heroic" acts – myths, mainly far away from Greece, e.g., in Hyperborea (far to the north of Thrace) and near the Polar area (after Plutarch). From the 12 labors, 6 were realized in Peloponnesus, 2 in the geographical region of Thrace and Caucasus, 1 in Crete, and 2 in Western Europe and NW Africa.

Regarding the topics of the labors, many of those as well as the lesser-known works of Heracles are related to greater and smaller interventions to the hydrological and hydrogeological regime. Such myths are those related to the Lernean Hydra, the Stymphalian birds and the Augean stables, the destruction of the flood prevention works of the Minyans in Kopais basin, the battle of the river Acheloos, and others. Additionally, the labor referring to the Nemean Lion is said to be related to the adjustment of the surface waters in the broader area of Nemea and the homonymous river.

5.1 Heracles and the Lernean Hydra

The myth of the extinguishing of the Lernean Hydra, which is the 2nd labor of Heracles, is one, which confirms the direct connection between the hydrogeological conditions of Lerna area and the myth.

A repulsive, snake-formed beast, named Lernean Hydra, used to live in the region of Argos (NE Peloponnesos), near the Lake Lerna. It had an enormous serpentine body, ending at several snaky tresses, with a head at each edge.

The beast's breath was poisonous and often Lernean Hydra would spit out fire while, even asleep, it used to destroy everything in the Argolic plain, including crops, trees, animals, and even people.

The myth of Heracles and Lernean Hydra could be interpreted through the hydrogeological conditions of the karstic springs system of Lerni [24], if we take into account the following:

- The area of Lerni is located a few kilometers south of Argos whereas, not far from it, two other karstic spring systems exist, that is the Kefalari spring to the north and the submarine spring of Anavalos, by Kiveri village to the south.
- The springs' altitude: The present altitudes of the springs vary. Specifically, Kefalari spring are at 24 m above the present sea level; Lerni springs are at 0.50–1.0 m a.s.l., while the group of Anavalos springs discharge under the present sea level, at a depth exceeding -5 m.
- The spring yield: The spring yield varies annually and seasonally, as naturally expected. So, in 1965, the mean annual yield of Kefalari spring was $\cong 80 \times 10^6$ m^3/year, the yield of the main Lerni spring was $\cong 60 \times 10^6$ m^3/year, whereas that of Anavalos springs was approx. 100×10^6 m^3/year.
- Despite the fact that the mean annual discharge at Lerni spring is relatively lower than the Kefalari one, dry up of the central spring has never been observed. The myth says that the "central head" of the Lernean Hydra was immortal.
- At Lerni, the discharge occurs at several points.

The number of points, from where the karstic aquifer is being discharged, as it is expected, differs and depends on three factors: (1) the season of the year, (2) the mean annual height of the atmospheric precipitation in the mountainous region of Arkadia and Korinthia provinces, and (3) the interval of the climatic period (whether the possible wet year is of 100, 500, or even 2,000 years returning period).

In this way, one may interpret the number of heads of Hydra as the different points of discharge of the karstic aquifer, which, as stated before, varies according to the existing climatic conditions during a longer or a shorter climatic period. According to what has been described, it is obvious that the myth of Lernean Hydra and Heracles is directly linked with the prehistoric and the present day hydrogeological conditions of Lerni springs.

The myth informs us on two more actions of Heracles and on a skill of Hydra, that is:

- Heracles faced Hydra with a sword, a cudgel, or stones, and
- When a head was cut, two new ones were sprout.

Heracles trying to exterminate Lernean Hydra started by decapitating, one by one, the beast's heads. The decapitation of each head which, in our opinion, represents a spring's discharge at a karstic point may become possible by the placement of a rock at the point where the water discharges, in order to prevent its exit or to force it to follow another route. It is well known, among geologists, that the karstified rock body, through which the underground water circulates on its way to the spring, represents a complex system of underground, intercommunicating erosion pipes or ducts. In addition, the tectonic discontinuities, even if the karstification is not very intense, are also permeable. So, if someone places a rock in front of the mouth of a karstic spring, the water will come out from two other or more points. That is the reason why in the place of a Hydra's head that Heracles cut, two others would sprout.

Concerning the sword, Heracles should have been using it in order to cut the thick vegetation that usually exists in front of a spring (swamp). Despite the fact that enormous quantities of water are pumped today by several drillings constructed in the area, it is easy to perceive that the vegetation of this hydrobiotope of prehistoric Lerni should have been quite rich, probably richer than that of the present period, especially during the period of the climatic optimum, when the sea level was higher than today. Heracles probably used the cudgel in order to smash the limestone at the spring mouth and Iolaus used the sickle in order to cut the vegetation and facilitate Heracles' way to the karstic springs, that is the heads or the central head.

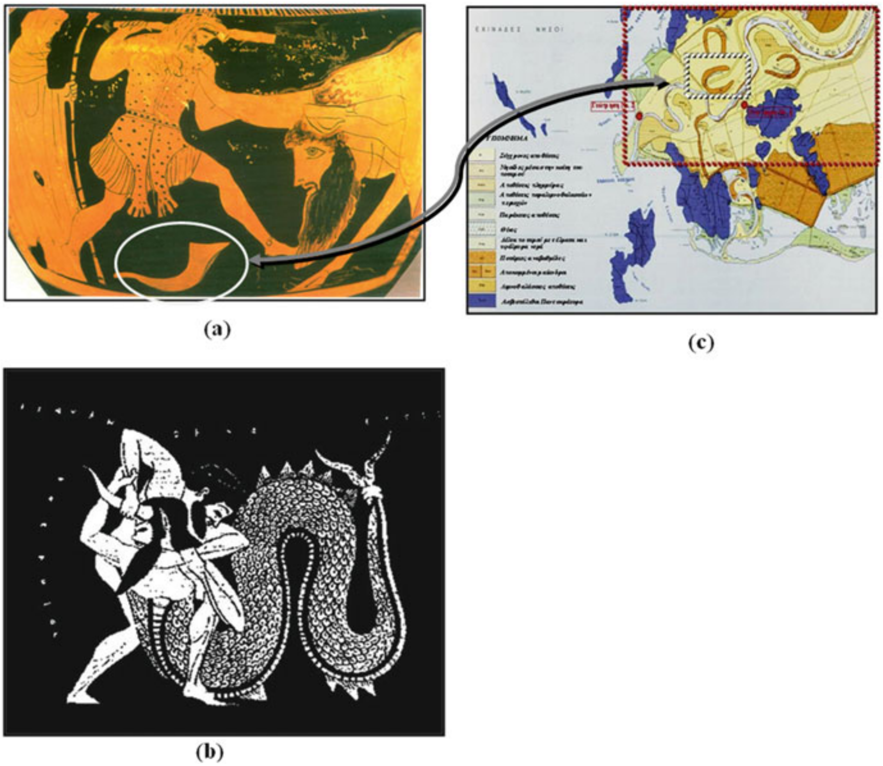
5.2 The Struggle of Heracles with Acheloos River

The myth mentions that Heracles visited the Aetoloakarnania province in Western Greece, after an invitation from Aetolians, in order to settle their issues with the Akarnans people, created by the changes that Acheloos River was causing on the riverbanks. At this point, it should be mentioned that the Acheloos River represented the borderline between them. During these times, *Oeneus*, the king of Calydon had a beautiful and dynamic daughter, *Deianeira*. River god Acheloos was in love with her. Acheloos had the ability to transform himself. He could transform to a dragon, a bull (Fig. 11a), or a reptile (Fig. 11b). These transformations had scared Deianeira.

When Heracles arrived, she fell in love with him and they decided to get married. Deianeira told Acheloos that she did not want him and then Acheloos challenged Heracles to a fight. In Fig. 11a, b the fight of Heracles with Acheloos is depicted. Heracles grabbed the bull from the head and cut one horn. In Fig. 11a, b, the reptile body of Acheloos can be seen, the head with the horns and the cut horn, which is called the "Amalthea's horn." If we relate the geomorphological characteristics of the river delta and the banks (Fig. 11c), we can see that the two figures match.

The Amalthea's horn is related to prosperity and wealth. In this case, it is known that the isolated lobes of river meanders are more fertile than the surrounding areas. What really happened in reality? Essentially, Heracles, constructing various hydraulic–geotechnical works on the riverbed, managed to accelerate the water flow in the Acheloos delta and turned away the water in order to avoid flooding, in order to create healthy conditions (minimize stagnant waters). Where do the works start? They start at the "head." Where is the "head" of a long river? It is the delta.

Heracles didn't carry out works at the upper river course since this part of the river was not causing issues between the Akarnans and the Aetolians. The problems existed in the lower parts of the river, in the delta region, where the riverbanks – for various reasons – were changing frequently causing problems to the societies of those times. In the Acheloos case, the myth is directly associated with the geo-environmental changes.



(a) (c) (b)

Fig. 11 (**a**, **b**) Representation of the struggle between Acheloos and Heracles, (**c**) geological map of the Acheloos river delta area and the cut-off meanders formed during the evolution of the delta

6 The Rivers in the Greek Mythology

6.1 Mythological Characteristics of the Rivers

According to the Greek Theogony, which was composed by the great poet *Hesiod*, rivers were "born" by the couple of Titans known as Oceanus and Tethys. These Titans gave birth to over three thousand rivers and to an equal number of Oceanides (sea nymphs, Fig. 12, fourth generation of Gods).

In order to honour the river gods, great artists had created many statues. Statues devoted to Kifissos River (Athens and Boeotia), Ilissos (Athens), Maiandros (Asia Minor), Alfeios and its tributary Kladeos (Peloponnese, Olympia), etc. The Acheloos River, for example, is depicted with a body of a bull and a human face (Fig. 11a), while at the southern end of the eastern pediment at the temple of Zeus in Olympia, Alfeios and Kladeos rivers are depicted as humans at the northern pediment. At the Parthenon pediment, there are human-like representations of the Kifissos and Ilissos Rivers.

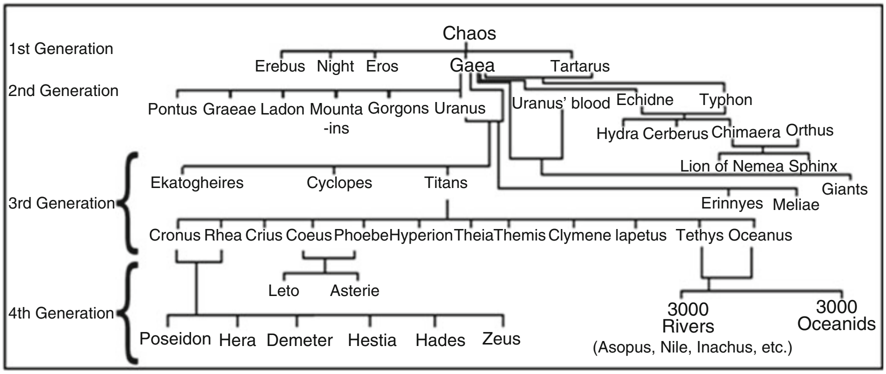


Fig. 12 The genealogy of the first four generations of Greek Gods, according to Hesiod

Some of the mythological characteristics of the rivers are the following:

- The rivers were considered as the Patriarchs of their regions, whereas, during puberty, the youngsters were dedicating their long hair.
- Most rivers are children of Titans Ocean and Tethys. Several exceptions exist, for example, river Evrotas that runs through the Sparta basin (Laconia, S. Peloponnesus). This exception is related to the special characteristics of the physical–geological evolution of Sparta basin.
- Another common characteristic is that they had the names of kings. For example, Inachos in Argolis, Acheloos in Aetoloakarnania, Maiandros in Ionia (Asia Minor), etc., whereas their name changed more than once, e.g., Acheloos was initially named Thoas, then Axenos, and then Thestios. Nile's name changed from Egypt and Maiandros to Anevainon, etc.
- Nowadays, rivers are considered sacred in India (e.g., the Ganges River).
- It is worth noting that since Homeric times, they refer to underground karstic rivers as well. There, the souls were going through a transition stage till the return to the surface. Such rivers are Acheron, Cocytus, Pyriphlegethon, and Styx that Plato refers to as a lake.
- Rivers are considered as the spawners of many islands or kings that gave their name to islands. For example, the Asopos River is mentioned as the spawner of Euboea, Aegina, Salamis, and other islands. The Acheloos River is considered the spawner of the Echinades islands, and Nile of Egypt.

The reason for the latter correlation is related either to the sea level changes due to climatic changes, probably because it was generally accepted that rivers are responsible for the rise of the sea level, or for the Delta development as in the case of the Nile (the part of the river in the Lower Egypt that developed after the Holocene Climatic Optimum). This belief, of course, is not scientifically valid. Today, we are aware that the sea-level rise is due to climatic–eustatic movements that relate to the glacier meltdown.

6.2 The Deification of Rivers

Rivers are the only physical geographical systems that have been deified. It is also remarkable that neither the Mountains nor Pontus (the oceans), the other large physical geographical systems, which belong to the 2nd generation, had been deified. Rivers have defined the cultural process of human societies, especially not only those who lived in the Aegean and Peri-Aegean regions, but also those who lived in regions surrounding the Eastern Mediterranean and the Black Sea. This is the reason that Greek Mythology also refers to rivers, which are located beyond the Greek territory like the Nile, Istrus (Danube), Eridanus, Borysthenes (Dnieper), Tanais (Don), and/or at the Hyperboreans Land.

In order to approach the rivers deification issue and attempt to answer this fundamental question, the following must be taken into account:

1. Residents were mainly choosing the deltas or the river banks and, in particular, the higher river terraces for their settlement, which were formed in the past and therefore provided safety in case of floods.
2. The deltas are not stable physical-geographical systems.
3. Water quantities and sediment supply were fluctuating, causing disasters with many casualties. It has been observed that even small climatic changes can cause major changes in riparian zones (e.g., [25]).

However, for us to comprehend the essential reasons, we must become more familiar with the periods that followed the beginning of the interglacial period that were the most important for the rivers' evolution. Therefore, according to what is known so far, it seems that three periods can be identified.

The first period extends from 18,000 BP up until right before 12,500 BP, that is, just before the start of the Younger Dryas cold stage (see Fig. 1, practically between 15,000 and 12,500 BP). The main reason for this is related to the fact that during that period a rapid rise in temperature occurred, which resulted at the rapid glaciers meltdown. In the greater Circum–Aegean region, this meltdown is related to the glaciers of the mountainous regions. Additionally, the rise in temperature was followed by an increase of rainfalls. Therefore, the residents who lived in the riparian zones, the river terraces, banks, and palaeo-deltas must have suffered from floods, cataclysms, and sediment supply deposits responsible for extensive disasters and casualties.

The second period starts with the end of the "Younger Dryas Period" (12,500–11,400 BP) and extends till the 6,000 BP, i.e., approximately until the start of the Holocene Climatic Optimum. During that period, the main physical–geological change is the rise of the sea level which had the following two major consequences: (a) the continuous flooding of the old Deltas (which was completed till 18,000 BP) and (b) the continuous formation of new islands due to climatic–eustatic movements and the inundation of the coastal areas of the older, greater palaeo-islands.

The third period, that starts at 6,000 BP and continues up to our days, is characterized by the stabilization of the sea level, the start of new Delta formations, and their gradual development.

Therefore, the question that arises is when and in which of these three periods the deification of rivers occurred. It is my personal belief that as the first period, i.e., 18,000–12,500 BP, is the most significant one then the deification of rivers must have occurred at that time. This view is also supported by the fact that many descriptions of the Greek Mythology refer to the same period, with the most representative myth being that of Acheloos River and the creation of the Echinades Islands at the Ionian Sea, between Kefalonia Island and continental Greece.

References

1. Vitaliano DB (1968) Geomythology. J Folklor Inst 5(1):5–30
2. Mariolakos ID (2010) The forgotten geographic and physical – oceanographic knowledge of the prehistoric Greeks. Bull Geol Soc Greece 43(1):92–104
3. Milankovitch M (1941) Kanon der Erdbestrahlung und seine Anwendung auf dem Eiszeitenproblem – Royal Serbian Sciences, Spec. Publ. 132, Section of mathematical and natural sciences, vol 33. Belgrade, 633 pp
4. Duff D (1993) Holme's principles of physical geology. Chapman & Hall, London, 791 pp
5. Mitropetrou EP (2012) The origins of the Greek geomythology through the cosmogonies, theogonies and the cycle of hercules. PhD thesis, University of Patras, 688 pp
6. Mariolakos I, Theocharis D (2002) Asopos River and the creation of Aegina and Salamis Islands (Saronic Gulf, Greece). A geomythological approach. In: Proceedings of "environmental dynamics and history in Mediterranean areas" Congress, Paris, 24–26 Apr 2002
7. Degens ET, Ross DA (1972) Chronology of the Black Sea over the last 25,000 years. Chem Geol 10:1–16
8. Lericolais G, Popescu I, Guichard F, Popescu S-M, Manolakakis L (2007) Water-level fluctuations in the Black Sea since the last glacial maximum. In: Yanko-Hombach V, Gilbert AS, Panin N, Dolukhanov PM (eds) The Black Sea flood question: changes in coastline, climate, and human settlement, part 4. Springer, Dordrecht, pp 437–452. ISBN 978-1-4020-4774-9
9. Major CO (1994) Late quaternary sedimentation in the Kerch area of the Black Sea shelf: response to sea level fluctuation. BA thesis, Wesleyan University, Middletown, CT, 116 pp
10. Ross DA, Degens ET, MacIlvaine J (1970) Black Sea: recent sedimentary history. Science 170:163–165
11. Ryan WBF, Pitman WC, Major CO, Shimkus K, Moskalenko V, Jones GA, Dimitrov P, Görür N, Sakinç M, Yüce H (1997) An abrupt drowning of the Black Sea shelf. Mar Geol 138:119–126
12. Aksu AE, Hiscott RN, Mudie PJ, Rochon A, Kaminski MA, Abrajano T, Yaar D (2002) Persistent holocene outflow from the Black Sea to the eastern Mediterranean contradicts Noah's flood hypothesis. GSA Today 12(5):4–10
13. Hiscott R, Aksu AE, Mudie PJ, Kaminski MA, Abrajano T, Yasar D, Rochon A (2002) The Marmara Sea gateway since ~16 Ky BP: non-catastrophic causes of paleoceanographic events in the Black Sea at 8.4 and 7.15 KY BP. In: The Black Sea flood question: changes in coastline, climate, and human settlement. Springer, Netherlands. doi:10.1007/978-1-4020-5302-3_5
14. Mariolakos I (2004) Geomythology. In: Birx JH (ed) Encyclopedia of anthropology, vol 3. New York, SAGE, pp. 1066–1071

15. Ryan W, Pitman WC (1998) Noah's flood: the new scientific discoveries about the event that changed history. Simon & Schuster, New York, p. 319
16. Mariolakos I (2002) Dardanos' flood and a dating attempt. In: Proceedings of panhell. Geogr. Congress, Thessaloniki, 3–6 Oct 2002, vol 1, pp 139–149 (in Greek)
17. Demek J, Kukla J (1969) Periglazialzone, löss und paläolithikum der tschechoslowakei. Czechoslovak Academy of Sciences, Institute of Geography, Brno, 156 pp
18. Major C, Ryan W, Lericolais G, Hajdas I (2002) Constraints on Black Sea outflow to the Sea of Marmara during the last glacial-interglacial transition. Mar Geol 190:19–34
19. Tchepalyga A (2003) Late glacial great flood in the Black Sea and Caspian Sea. G.S.A. abstracts of the session No 187: "Noah's flood" and the late quaternary geological and archaeological history of the Black Sea and adjacent basins
20. Knauss J (1991) Mykenische Talsperren in Arkadien und Böotien – Historische Talsperren, 2, Deutsche Verband f. Wasserwirtschaft und Kulturban e.V. Vrlg. K. Wittwer, Stuttgart, pp 19–39
21. Knauss J (1984) Die Wasserbauten der Minyer in der Kopais – Die älteste Flussregulierung Europas (Kopais 1) – Wasserbau und Wasserwirtschaft, Nr. 50. Technische Universitaet Muenchen
22. Angelakis AN, Savvakis YM, Charalambakis G (2006) Minoan aqueducts: a pioneering technology. In: Angelakis AN, Koutsoyiannis D (eds) IWA 1st international symposium on water and wastewater technologies in ancient civilizations, Iraklio, Greece, 28–30 October, pp 423–429
23. Angelakis AN, Spyridakis SV (1996) Wastewater management in Minoan times. In: Proceedings of the meeting on protection and restoration of environment, Chania, Greece, 28–30 Aug, pp 549–558
24. Mariolakos I (1998) The geomythological geotope of Lerni Springs (Argolis, Greece). Geol Balc 28(3–4):101–108
25. Seavy NE, Gardali T, Golet GH, Griggs FT, Howell CA, Kelsey R, Small SL, Viers JH, Weigand JF (2009) Why climate change makes riparian restoration more important than ever: recommendations for practice and research. Ecol Restor 27(3):330–338
26. Apollodorus (1999) (2nd cent. B.C.) Apanta 1, bibliotheca, Book 1, Ch. 7. Kaktos, Athens (English translation by Sir James George Frazer, Loeb Classical Library Volumes 121 & 122 (1921) Harvard University Press, Cambridge; William Heinemann, London. http://www.theoi.com/Text/Apollodorus1.html#7)
27. Siculus D. Historical library, vol 47, pp 3–5 and vol 48, pp 2–3
28. Strabo. Geography (Pelopponnesus). Book 8, Chap. 3, Sect. 19

Natural Processes Versus Human Impacts During the Last Century: A Case Study of the Aliakmon River Delta

Michael Styllas

Abstract The Aliakmon River flows down from the northwestern mountains of Greece and is one of the largest fluvial systems in the Greek territory. Basin climate and geology favour the high rates of sediment production and transport and, consequently, the formation of an extensive (9.2% of basin area) bird-foot Holocene delta. Three phases (A, B and C) of human impacts over the past 90 years have caused pronounced changes on the natural evolution of the delta. During Phases A and B, a 50% increase of deltaic sedimentation rates in relation to Holocene pre-anthropogenic rates and an enrichment of deltaic deposits with heavy minerals occurred. Phase C, characterised by damming, increasing agricultural and industrial activities and population growth, resulted in 90% decrease in sedimentation rates compared to Phase B, a regulated hydrological regime with high electrical conductivity and nutrient concentrations of surface water, enhanced erosion of river channel and deltaic deposits and degradation of habitats along the lower Aliakmon River delta. Future climate scenarios and increasing environmental pressures are not compatible with current water use strategy and, given the vulnerability of the system (reservoirs and delta) to projected climate trends, stress for a new strategic natural resource management plan.

Keywords Aliakmon River, Deltaic sedimentation, Human impact, Natural resource management, Water quality

Contents

M. Styllas (✉)
GEOSERVICE Ltd, Eirinis 15 Street, Panorama, 55236 Thessaloniki, Greece
e-mail: mstyllas@gmail.com

N. Skoulikidis et al. (eds.), *The Rivers of Greece: Evolution, Current Status and Perspectives*, Hdb Env Chem (2018) 59: 31–50, DOI 10.1007/698_2017_470,

1 Introduction

River deltas are linked to the evolution of many civilisations since the Stone Age. In many cases, coastlines have been formed by the interaction between fluvial and marine processes, while deltaic plains are the areas where agriculture was initially established. Since the early Bronze Age, when Sumerian, Babylonian and Assyrian empires evolved in the lower Tigris–Euphrates River, river deltas have been reclaimed and modified by humans. A detailed understanding of the local biogeochemical processes driving deltaic formation and evolution is essential in assessing the extent and magnitude of human impacts as well as in order to provide estimates of natural resource availability and proper future planning.

The Aliakmon River, which is the second largest fluvial system exclusively laying in Greece, exhibits a strategic status in terms of water resource and deltaic plain management. It is a mountainous river originating in northwestern Greece, discharging water and sediment into the Thermaikos Gulf (Fig. 1). Since the 1930s, different types of human impacts have disturbed its deltaic natural evolution, thus turning the lower Aliakmon River from natural to a human-controlled system with adverse consequences on water and sediment regimes.

The natural evolution of the Aliakmon River delta during the last century is presented here through estimates of selected quality and quantity parameters for water and sediment. A number of anthropogenic modifications on the watershed and delta for the past 90 years are cited. Natural (unregulated) fluvial water and sediment discharge estimates are derived through numerical modelling, as data sequences do not extend prior to 1925, when the river is considered to be undisturbed. On the contrary, the effects of anthropogenic pressures on water and sediment budgets are described by means of sample analyses, as the river lacks a monitoring network. Comparison of specific parameters between natural and disturbed periods provides useful conclusions about the present-day functionality of the lower course

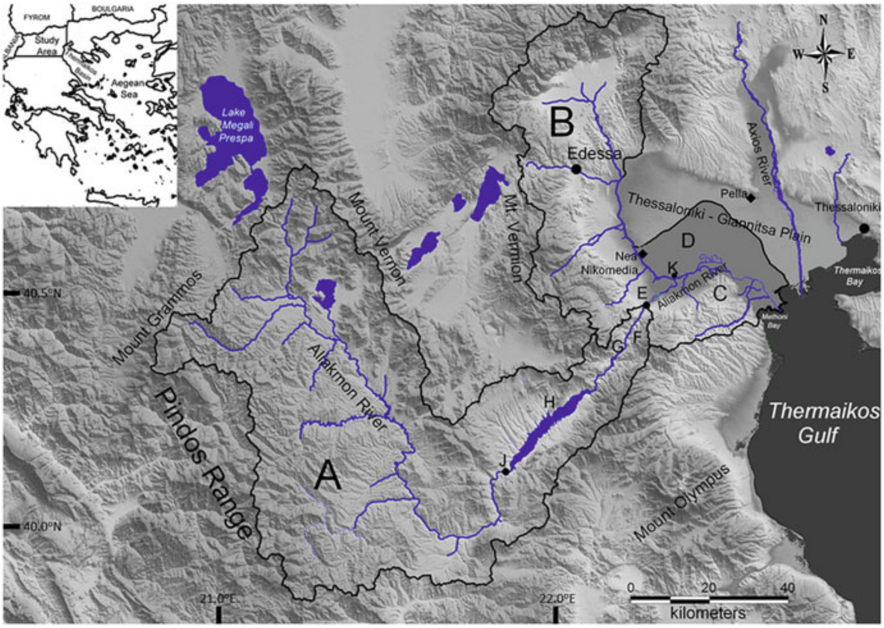


Fig. 1 General setting of the study area with all geographical elements cited in the study. A: Aliakmon River watershed with major subbasins (total area, 6,100 km^2). B: Peripheral Canal watershed with major subbasins (added to Aliakmon River basin in the 1930s, following Phase A of human interventions; total area, 2,223 km^2). C: Pieria subbasins (contribute water and sediment to Aliakmon River delta during the Holocene; total area, 453 km^2). D: Aliakmon River Holocene delta (defined from satellite elevation and topographical data and geomorphological land observations; total area, 567 km^2). Total drainage area of A, B, C and D sums to 9,343 km^2, the present-day surface area of Aliakmon River basin. E: Aliakmon River Holocene delta apex (the river's exit to the valley). F: Asomata reservoir. G: Sfikia reservoir. H: Polyfyto reservoir. J: Agios Ilarionas dam. K: Junction of Aliakmon River with Peripheral Canal. The Neolithic settlement of Nea Nikomedeia and the capital of Macedonian Empire Pella are also shown

of the Aliakmon River delta. The present study aims in providing a comprehensive review of the current environmental status and also of the environmental threats of Aliakmon River delta under climate change scenarios. The work presented here is considered as the basis that will comprise a useful tool for the political initiatives and future planning of Greece's most important fluvial system.

2 Thermaikos Basin

2.1 Geological Evolution and Sedimentary Depositional Regime

The Thermaikos basin is strongly related with the geological evolution of Aliakmon River basin. The thickness of Cenozoic deposits in the Thessaloniki–Giannitsa plain and the Thermaikos Gulf is 3,000 m [1]. Within the Thermaikos basin, the existence of molasses and lignite layers indicates the changing nature of depositional environments during the first cycle of basin subsidence, which is dated between Upper Oligocene and Lower Miocene [2]. The early Neogene phase of the Thermaikos basin subsidence is related to the opening of the Aegean back-arc basin that resulted from the southward retreat of the Hellenic subduction zone [3]. From Middle to Upper Miocene, the Thermaikos basin entered a period of sea-level regression that was characterised by extensive formation of red oxidised soil layers. A second phase of tectonic subsidence and deposition of lacustrine and shallow-marine sediments began in Early Pliocene. Continuous faulting along basin margins associated with the dextral motion of the western end of North Anatolian Fault [4] resulted to the deposition of volcanic tuffs of trachyandesite (felsic) composition [2].

Between Pliocene and Pleistocene, the basin entered another period of sea-level regression due to intense uplift that resulted in thermal spring activity and deposition of travertines along the basin margins. During the Pleistocene and the Holocene, sediment transport regime from the marginal rivers towards Thermaikos basin was mainly defined from eustatic and isostatic movements.

2.2 *Oceanographic Setting*

The Thermaikos Gulf (Fig. 1) is a semi-enclosed embayment at the NW part of the Aegean Sea. The oceanography of the gulf is characterised by low-energy wind, wave and tide regimes. Prevailing winds generally blow from north-northwestern directions, their velocities exceeding 15 m/s less than 1% of the year [5]. During the winter, northerly wind outbreaks of gale force (with velocities of 20 m/s) known as 'Vardaris' are funnelled to the gulf mainly through the river valley of the Axios River, resulting in abrupt surface water temperature lowering and cyclonic circulation pattern along the western coast of the gulf. Northern winds tend to be weaker and less frequent during the summer months [6].

Mean annual significant wave height is less than 0.5 m, and significant wave heights exceeding 3 m have been recorded at a frequency of 1%. Tidal range across the gulf varies between 30 cm at mean spring tides and 5 cm at mean neap tides [5]. In this relatively calm and tideless environment, the formation of the Aliakmon delta has been largely determined by the interaction between water/sediment discharge and wave action, the wave power not exceeding 30 W/m^2 with maxima observed between mid-spring and mid-autumn [7] coinciding with the period of low water and sediment discharge of the Aliakmon River.

3 Aliakmon River Watershed and Delta: Physical Characteristics

3.1 Basin Geology

The Aliakmon River originates on the northeastern side of the Pindus Range in continental Greece. The river's basin has a wave-shaped form and a surface area of 6,100 km^2 at the exit to the plain of Thessaloniki–Giannitsa (Fig. 1, E). The highest altitude of Aliakmon River watershed is located on the summit of Grammos Mountain (2,520 m.a.s.l.). Average drainage basin elevation is 836 m.a.s.l. with 32% of the total basin area being confined between 600 and 800 m.a.s.l., while average relief ratio is 1.7 10^{-2} [8].

The Aliakmon River length is 310 km along which the river drains four geotectonic zones, each of them corresponding to a different paleogeographical setting. Processes, such as uplift, erosion and intrabasin deposition during two major cycles (2.1), have created a variable basin lithological composition: felsic rocks 14.5%, mafic rocks 9.2%, volcanic rocks 0%, carbonates 15.7%, flysch–molasse 29.6% and Neogene and Quaternary sediments 31% [9].

The first erosion–deposition cycle took place during the Tertiary (Upper Oligocene and Lower Miocene) and resulted in excessive intrabasin and down-valley deposition of Neogene terrestrial, fluvial and lacustrine deposits. The second cycle of erosion is placed along the Pliocene–Pleistocene boundary and resulted in the incision of the Aliakmon gorge and the formation of the river's deltaic plain that was gradually silted up during the Quaternary.

3.2 Basin Climate

The Aliakmon River basin climate is characterised as 'continental' along its main watershed becoming 'Mediterranean' towards the deltaic plain [7]. Annual average values of main hydrological parameters demonstrate a west-east gradient. Higher values to the west result from the orographic effect of Pindus Range to the wet fronts arriving from the Adriatic Sea, gradually decreasing towards the eastern part of the watershed.

Data collected between 1963 and 1999 in the Agios Ilarionas dam (Fig. 1, J), a location representing 82% (5,005 km^2) of basin area (Fig. 1, A), provide annual average values of precipitation 764 mm, air temperature 12.2°C, evaporation and transpiration 435.4 mm (representing 56.2% of precipitation) and total runoff 339.4 mm/y representing the remaining 43.8% of total annual precipitation [10], while the entire watershed precipitation and temperature annual averages are 750 mm and 16.5°C, respectively [11].

3.3 River Discharge

Limited measurements at the river exit to the valley during 1930s indicate mean annual discharge values of 95 m^3/s [12], whereas hydrological model output estimates spanning a 40-year period (1960–2000) data reanalysis from the entire basin provide a better estimated mean annual discharge of 70.6 m^3/s [10], with minimum and maximum average monthly discharge values ranging between 21 and 137 m^3/s, respectively [11]. Measured discharge values at the river mouth during the 1997–1998 METROMED project [13] averaged 34 m^3/s, illustrating the effects of anthropogenic impacts on river Aliakmon's discharge. During periods characterised by rain on snow events, the Aliakmon River has exhibited discharge values higher than 3,200 m^3/s, resulting to frequent flooding of the lowlands since the ancient times [12].

Spectral analyses of monthly precipitation and discharge suggest that river discharge annual distribution peaks occur during the winter months (December–March) in contrast to double peaks of precipitation maxima occurring during October–December and April–June, respectively (Fig. 2). This discrepancy is explained by the high infiltration rates that result from basin lithology (75% of Aliakmon basin formations are either soft sediments or carbonates), from sparse vegetation and basin climate, with dry summers characterised by high evaporation rates (56% of total

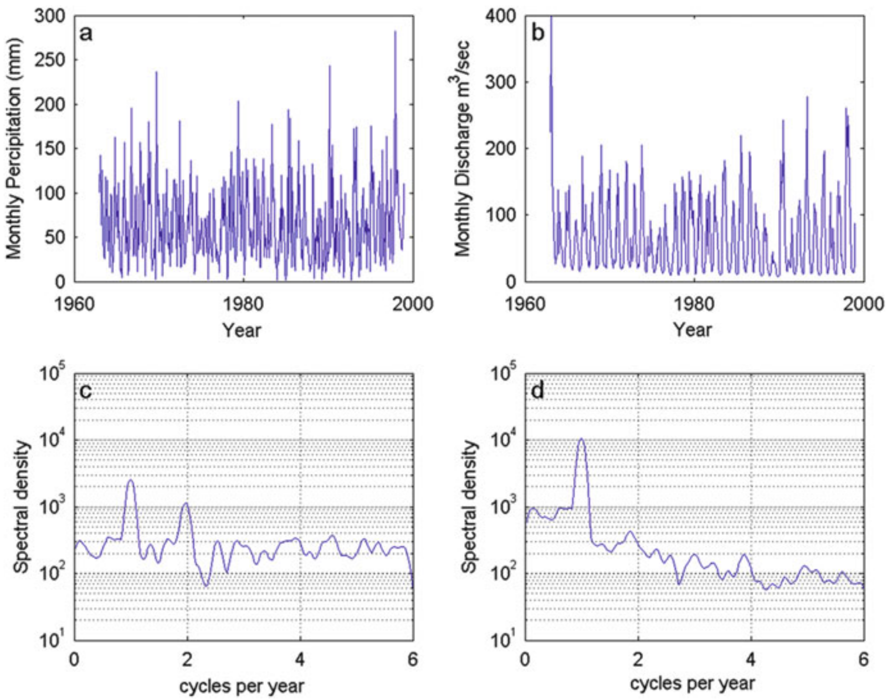


Fig. 2 Time series of (**a**) precipitation and (**b**) discharge monthly average values of Aliakmon River at Agios Ilarionas (Fig. 1, J) gauging station and (**c**, **d**) their associated spectra

precipitation) and thick unsaturated soils (high denudation). Such hydrological conditions require large amounts of precipitated water to provide adequate surface runoff during the autumn rain period, resulting to delay of the high discharge period. Increased discharge values are observed at the end of the autumn rain period (December) peaking during early spring (March), the maxima explained by snowmelting, rain on snow events and saturated soils.

The chemistry of Aliakmon River surface water is determined by basin climate and basin lithology as both factors control the type of weathering. The geochemical signal of river water is dominated by calcium (Ca^{2+} 38.5 mg/l), magnesium (Mg^{2+} 31.3 mg/l), silica (SiO_2 11.4 mg/l) and chloride (Cl^- 5.6 mg/l) contents. While the main portion of major ions is related to weathering of recent (Neogene and Quaternary) sediments, magnesium and silica are also derived from mafic rock weathering [9, 14].

3.4 Deltaic Stratigraphy

In sequence stratigraphic terms, the Holocene delta of Aliakmon River is divided into three systems tracts: (1) LST, a low-stand systems tract of variable thickness, composed of fluvial gravels and sands (alluvial fan) of the Late Pleistocene, as well as from red oxidised clays (alluvial plain); (2) TST, a relatively thin (2–8 m) transgressive systems tract composed of fluvial channel sands, overlain by a thin transgressive sand bed of coastal origin, characterised by fining upward (FU) grain-size trends that indicate a phase of sea-level transgression; and (3) HST, high-stand systems tracts (5–35 m), constituted by a variety of stratigraphic units, stacking patterns and depositional environments (fluvial channel, levee channel, coastal lagoon, marsh, delta front and floodplain); characterised by coarsening upward (CU) sequences, representing both aggradational (sea-level rise rate = sedimentation rate) and progradational (sea-level rise rate $<$ sedimentation rate) facies; and dominated by the presence of three distinct progradational wedges associated with climatic (high sedimentation rates) and/or eustatic (still stands, tectonic uplift) oscillations [15].

The estimated volume of Holocene deltaic deposits derived from the underground mapping of TST and HST from drill profiles by using a measured (100 samples) average-specific weight value of $\gamma = 1.49$ g/cm^3 is 26.3×10^9 m^3.

3.5 Deltaic Sedimentation

For the purposes of the current study, natural deltaic sedimentation rates have been estimated by the following methods: (1) From application of an empirical power law function on annual water and sediment discharge data measured by the Greek Public Power Corporation at the Agios Ilarionas dam (Fig. 1, J) and extended by linear interpolation to cover the entire drainage basin (Fig. 1, A + C + D). The results suggest average annual estimates at the present-day river mouth of 6.75×10^6 t/y.

(2) From estimates of Aliakmon Holocene delta (Fig. 1, D) accommodation space derived from drill data that penetrated the Pleistocene–Holocene boundary (Sect. 3.4). Drill data suggest average sedimentation rates of 2.5 m/ky in agreement with the findings of [8] that employed geophysical methods and estimated the bottomset and foreset deposition to be 0.5 and 3.0 m/ky, respectively. Such sediment accumulation rate values, together with the underground mapping of the Holocene (10 ky BP) lower boundary, marked by transgressive systems track deposits (Sect. 3.4), provide an average annual Holocene sedimentation rate of 6.52×10^6 t/y. (3) From estimates of the Holocene deltaic sequence thickness at the Aliakmon River mouth, derived from seismic profiling and quantification of shoreline and bathymetric changes from digitised hydrographic maps between 1850 and 1916, a period lacking substantial human impacts on watersheds and delta. This method provided natural sediment discharge values at the present-day mouth of 6.63×10^6 t/y [16]. All three methods are in general agreement suggesting that average natural sediment discharge of Aliakmon River at its present-day mouth is 6.6×10^6 t/y. The relatively high annual sediment yields of 462 t/km^2/y [8], combined with high flood discharge values (Sect. 3.3) and low wave energy (Sect. 2.2), have resulted to the formation of a bird-foot delta during the Holocene, still evident from high-resolution topographic data (Fig. 3).

The geochemical composition of Aliakmon River deltaic sediments is defined by weathering processes on the watershed and biochemical processes that follow sediment deposition in a variety of environments along the delta. On average, Aliakmon River deltaic sediments contain low organic matter concentrations (2.4%), while higher concentrations (5%) are observed within lagoonal and marsh environments [15].

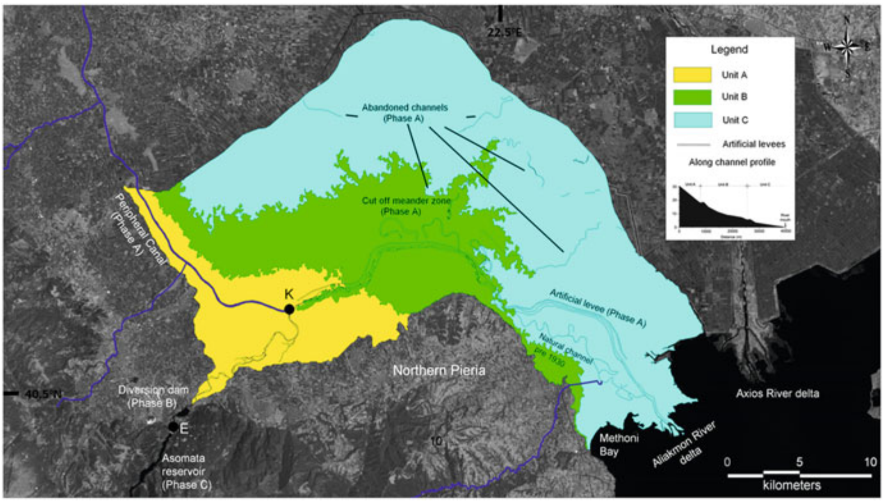


Fig. 3 Digital elevation data (SRTM 90) superimposed on satellite imagery (LANDSAT 97) illustrating the Holocene delta of Aliakmon River with its most prominent morphological features and human impacts. Extensive cut-off of the river's meander zone was realised in the 1930s for flood control. Abandoned channels, former lobes and the bird-foot shape of the Holocene delta are also evident

Calcium carbonate ($CaCO_3$) average concentration is 13% and reflects basin lithology and/or biological activity confined within delta marshes and back barrier lagoons, while observed down-core increase of $CaCO_3$ is attributed either to a decalcification process that results from a transient drop of pore water pH or to temporal decrease of biological activity.

Deltaic sediment sand fraction is rich in minerals containing silica (quartz 48.5%), calcium (epidote 13.8%, augite 2% and plagioclase 1.6%), magnesium (hornblende 5.8%) and potassium (muscovite 7.4%, feldspar 3.7%). Considerable amounts of underdetermined rock fragments (11.5%) and traces of biogenic silica (2.15%) are also present [1]. Clay mineralogy of river Aliakmon deltaic deposits is dominated by chlorite and kaolinite (up to 30%) that result from mechanical weathering of mafic rock formations, while smectite is more abundant in offshore (Thermaikos Gulf) locations [11].

3.6 Deltaic Evolution

The Aliakmon River delta plain has an area of 567 km^2. Holocene delta morphology derived from elevation and sedimentological data analyses is partitioned in three major units: The delta apex (12.5–30 m.a.s.l.) covering an area of 88.2 km^2 (15%) is registered as morphological Unit A and characterised by coarse-grained deposits (boulders, gravel and sand). Morphological Unit B is made up of sandy deposits of various origins (delta front, river channel, channel levee and floodplain) and covers an area of 151.8 km^2 (26%), its elevation bounded between 2.5 and 12.5 m.a.s.l., while morphological Unit C is characterised by fine-grained sediments, covers the majority (59%) of the Holocene delta surface (348 km^2) and is bounded between the 0 and 2.5 m.a.s.l. elevation contours (Fig. 3).

The subaqueous part (delta front and prodelta) of Aliakmon River modern delta interfingers with the prodeltas of the closely located Axios River and other minor rivers (Gallikos, Loudias), extending almost 50 km to the southeast and characterised by smooth gradients. Altogether, this complex system covers an area of 51,000 km^2 of the Thermaikos Gulf continental shelf, down to a depth of 200 m.b.s.l. [17].

The stratigraphy and morphology of the Aliakmon River delta are indicative of a mountainous river, characterised by high sediment transport rates and a rapid growing delta. The delta's central lobe rapid progradation that followed the stabilisation of sea level during mid-Holocene is partly responsible for the siltation and abandonment of the ancient harbour of Pella (Fig. 1) at 2,350 y BP [18]. Late Holocene stages of deltaic evolution still evident in aerial and satellite photos and topographic maps are characterised by delta progradation, flooding, lobe switching, frequent abandonment of active channels, high sinuosity meandering and avulsion of the main channel to the southeast, a result of the interplay between climatic, oceanographic and tectonic forcing. Further studies are required to define the later evolution stages of Aliakmon River delta.

4 Human Impacts on the Aliakmon River

Human presence along the western part of Aliakmon river delta dates back to the early Neolithic, in a settlement close to the present-day village of Nea Nikomedeia [19]. The excavated Neolithic settlement is considered as the oldest farming village in Greece [20]. Except farming, other interventions on the natural evolution of the Aliakmon River had not been realised until the beginning of the twentieth century. Major political events in 1922 resulted to the migration of more than 150,000 Greek refugees from Asia Minor to Northern Greece, forcing the Greek government to reclaim the plain of Thessaloniki–Giannitsa by means of hydraulic works along the channels of the main rivers, both for social health condition improvement and for the initiation of systematic agriculture in the area. Based on their timing, expanse and type, human impacts along the Aliakmon River watershed and delta are divided into three major phases.

4.1 Phase A (1925–1934)

The primary goal of the 'Reclamation Project of Thessaloniki–Giannitsa Plain' was the drainage of swamps and lakes through canalisation of rivers and streams draining the eastern (Mount Vermion) part and discharged directly onto the plain. These rivers were canalised along their lower courses as their water and sediment loads were diverted into a trapezoidal concrete drainage channel, the Peripheral Canal (Fig. 3), which was constructed between 1925 and 1930. The Peripheral Canal joins the Aliakmon River approximately 40 km upstream of its present-day mouth (Figs. 1 and 3, K). Along with drainage of floodplains, artificial levees were constructed to prevent flooding, thus protecting the newly established agricultural areas. In the case of the Aliakmon River, an artificial levee 38.5 km long and 6 m high was constructed along the left (north) bank of the channel and resulted to the cut-off of the river's meander zone (Fig. 3).

In contrast to the Aliakmon channel length reduction, the addition of the Peripheral Canal basin resulted to a drainage area increase of 2,223 km^2 but of different lithological composition (felsic rocks 4.9%, mafic rocks 12.3%, volcanic rocks 9.6%, carbonates 34.5%, flysch–molasse 8.1%, Neogene and Quaternary sediments 23.7%) most notably marked by the presence of Almopia volcanics.

4.2 Phase B (1934–1974)

Following drainage and protection of delta plain from flooding, reclamation of the land and initiation of systematic agriculture, the need for an irrigation network became apparent. Even though small-scale interventions never stopped after the termination of Phase A, it was not until the mid-1950s that the second phase of human impacts was more evident in the area. Hydraulic works included the construction of a diversion dam and water reservoir at the exit of the Aliakmon River gorge (Fig. 3) as well as of an extensive irrigation network (started in 1963 and completed in 1988) along the river's deltaic plain. The irrigation network expands beyond the boundaries of the Aliakmon River Holocene delta spanning an area of 774 km^2, nearly 70% of which is supplied with water from Aliakmon River [21].

In addition to the irrigation network, numerous roads and artificial sea walls along the coastal zone were constructed. The construction of the latter took place due to the continuous subsidence of the drained areas, a result of prodelta fine-grained deposit compaction. Even though Phase B had no direct impact on deltaic sedimentation and water quality, it did raise water demand issues and the need for additional human interventions that affected the evolution of the Aliakmon delta.

4.3 *Phase C (1974–Today)*

The fact that 95% of Axios River watershed belongs to FYROM and the amount of water reaching Greece and its delta is regulated from the neighbouring country raised the issue of the construction of a succession of hydroelectric dams along the Aliakmon River. The Greek government decided to construct the first of the four reservoirs in Polyfyto (Fig. 1, H). Construction began in 1970 and operation of the hydroelectric power plant (HEP) in 1974. In addition to Polyfyto (reservoir capacity, 1.937×10^6 m^3) HEP, two more HEPs were completed in 1985, the HEPs of Sfikia (Fig. 1, G; reservoir capacity, 103×10^6 m^3) and Asomata (Fig. 1, F; reservoir capacity, 53×10^6 m^3), the three of them covering a total area of 81 km^2 [22]. At present, the fourth HEP of Agios Ilarionas (reservoir capacity, 520×10^6 m^3) has started to operate, while the Greek Ministry of Environment, Energy and Climate Change and the Public Power Corporation are opting to construct additional minor reservoirs for electrical power generation and irrigation needs.

The construction of dams divided the river into two parts: upper Aliakmon (upstream of the dams) and lower Aliakmon (downstream of the dams). Phase C was also characterised by significant increases of fertiliser and pesticide use and urban and industrial activities (fruit and vegetable canning units) along the Peripheral Canal and by sand mining for highway construction, all leading to significant landscape and functionality changes of Aliakmon fluvial system.

5 From a Natural to a Human-Controlled System

The three phases of human impacts on river Aliakmon watershed and delta have had a direct impact on deltaic evolution. Despite the fact that Aliakmon is one of Greece's most important fluvial systems, it is lacking a thorough description of its present-day condition. The man-caused alterations of sedimentary and water regimes associated with these impacts stress for an accurate account of the human impact effects on the delta.

5.1 Effects on Water Regime

Isolation of lower Aliakmon from its headwaters and water regulation caused by damming, together with an increase of population and industrial activities in the vicinity of Peripheral Canal, had a significant impact on the Aliakmon River flow regime and ecological quality. The upper part of the river transports water, sediment and pollutants (domestic effluents and fertilisers) into the Polyfyto reservoir. The estimated annual organic load of the lake is 2,000 tonnes (BOD units), approximately 80% of which is transported by the Aliakmon River [23]. The Sfikia and Asomata reservoirs receive water from the Polyfyto reservoir but are largely unaffected and free of any sources of pollution, as indicated from the application of ecological monitoring (benthic macroinvertebrate abundance, BMWP taxa and values of biotic scores) at downstream locations from the Asomata reservoir [24].

The lower part of the Aliakmon River, below the reservoir of Asomata, flows through its Holocene channel belt bounded by an artificial levee along its left bank (Sect. 4.1). Released water discharge is considerably lower than natural, as indicated by the comparison of modelled and released discharge data at the Asomata reservoir between 1986 and 1999 (Fig. 4a), together with a shift of the high discharge period from spring to summer (Fig. 4b). On a daily basis, the Aliakmon River flow had been largely controlled by the operational needs of Asomata HEP and irrigation needs, hydrologically expressed as daily freshwater pulses [6]. This phenomenon ceased in 2008 with the construction of an additional reservoir downstream of Asomata HEP, which regulates a steady flow for the Aliakmon River with minimal discharge of 4.5 m^3/s (Aliakmon River Hydro Group – PPC, personal communication).

Aliakmon River water geochemical composition close to its present-day mouth [25] differs from the river's upper part (Sect. 3.3). Calcium concentrations are higher by a factor of 2 (Ca^{2+} 66 mg/l), an increase explained by the fact that the watershed between the Polyfyto and Asomata reservoirs is composed of carbonate rocks. Magnesium concentrations are similar to the upper part (Mg^{2+} 29 mg/l), while increased chloride concentrations result from polluted water (urban sewage) transferred to the lower Aliakmon River through the Polyfyto reservoir (Cl^{-} 34 mg/l at the exit of Asomata reservoir) and Peripheral Canal (Cl^{-} 40 mg/l close to the river mouth). As a result, electrical conductivity of the Aliakmon River surface water along its lower course exceeds the EU-suggested levels (250 mS) for drinking water (EU Council Directive for Drinking Water 80/778/EC).

The intersection of Aliakmon River with Peripheral Canal (Figs. 1 and 3, K) comprises a significant point of water quality degradation. Water of 'poor' quality from Peripheral Canal [24], which receives substantial loads of effluents and other compounds from urban (sewage and detergents), agricultural (fertilisers and pesticides) and industrial (fruit and vegetable cannery) sources, is transported to the

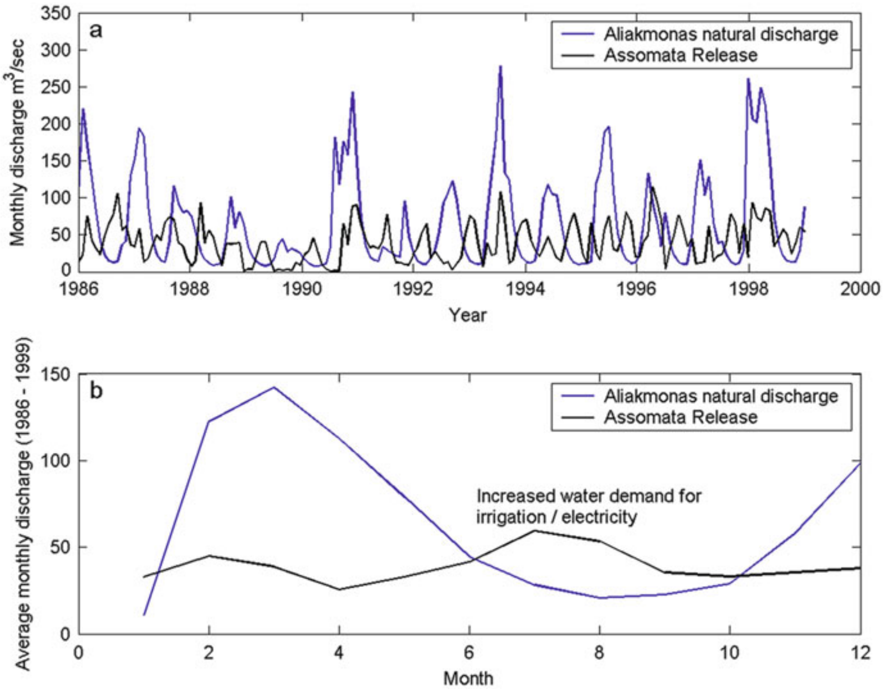


Fig. 4 The effects of human impact on the Aliakmon River hydrological cycle. (**a**) Reduced discharge at the river's exit to the delta plain (Asomata HEP). (**b**) Temporal displacement of the river's high discharge period, a result of irrigation and electricity demand

Aliakmon River. During low-flow season (September 1995), the Peripheral Canal nutrient (NO_2-N, NO_3-N and PO_4-N) concentrations exceeded EU levels. A strong seasonal signal of the Aliakmon River water quality deterioration close to its mouth is evident during summer and autumn months, associated with the intensification of agricultural and industrial activities as well as the abstraction of water for irrigation [24].

In the absence of a monitoring network, nutrient long-term trends, as derived from sparse data mainly referring to the autumn (low discharge) season, indicate a remarkable increase in total nitrates and phosphates and a subsequent degradation of the river's water quality through time. During Phase B (1969), a sampling field campaign highlighted the 'excellent' quality of Aliakmon and Axios Rivers' surface waters [1]. Seventeen years later (1985), the Aristotle University of Thessaloniki (Civil Engineering Department) conducted a study concerning the 'Water Quality of Thermaikos Bay' and pointed out potential eutrophication issues for Aliakmon River surface waters as total nitrate concentrations were considerably high (Table 1). Twenty years later, the sampling during 2005 indicated total nitrate and phosphorus concentrations to be considerably higher [25] than the previous decade.

Table 1 Long-term nutrient concentrations (mg/l) at the Aliakmon River delta

Date	NO_2-N	NO_3-N	PO_4-P
October 1985[a]	0.02	6.18	0.30
October 2005[b]	0.006	7.92	1.82

[a]After AUTH, Civil Engineering Department (1987)
[b]After Ilias et al. [25]

Increase in nitrate results from agricultural runoff and waste waters, as 1985 and 2005 values exceed EU values (NO_3-N 5.6 mg/l) for drinking water (EU Council Directive for Drinking Water 80/778/EC). Phosphorus was 6–12 times higher than previous (1985, 1995) measurements, an increase probably attributed to fertilisers, increase of industrial units along the Peripheral Canal and increase of population as urban (household) pollution contributes large amounts of phosphorous into surface water bodies.

Periods of high agricultural and industrial activity (summer–autumn) coincide with periods of low discharge and are characterised by very low water quality [24, 25] with immediate impacts on delta flora (degradation of riparian vegetation) and fauna (reduction of aquatic life habitats). As a consequence, the human-controlled lower threshold of 4.5 m^3/s appears inadequate to maintain the river's purification capacity, so locations close to the river mouth exceed EU levels for conductivity, nitrates and phosphates. Poor environmental quality at the Aliakmon River mouth is expected to have an immediate impact on local fishery and mussel farming units when 80% of Greece's mussel production units are located off the mouths of the Axios and Aliakmon Rivers. Moreover, according to the Aliakmon Hydropower Group, the four major HEPs on the Aliakmon River upper course account for an average of 4% of the total generated electric power in Greece, while their reservoirs have a total capacity of 1.937×10^9 m^3, the equivalent of the river's annual natural discharge.

The main concern for the Aliakmon River delta rises from the ever-increasing demand for water consumption. The Aliakmon River is the main water contributor for the city of Thessaloniki with average annual water volume of 88.3×10^6 m^3. Annual irrigation needs of more than 750 km^2 of land across the plain of Thessaloniki–Giannitsa require 520×10^6 m^3 of water. The Polyfyto reservoir contributes annually 35×10^6 m^3 to irrigation needs in locations upstream of the reservoir, yet another 65×10^6 m^3 is used for cooling four thermoelectric power plants located in close proximity to the reservoir. In addition, the Aliakmon River lower part contributes water for industrial needs with 32×10^6 m^3 annually, while 14×10^6 m^3 is the river's minimal annual discharge, less than 2% of the volume consumed for all other (irrigation, industrial, drinking, etc.) needs combined. In the near future, the Aliakmon River is planned to cover the irrigation needs of its right bank (south) agricultural area (340 km^2), to provide water to the Pella irrigation network (50 km^2), while the annual water transfer to the city of Thessaloniki calls for a rise up to 220×10^6 m^3.

Despite sparse sampling and non-existing monitoring that prohibit quantitative conclusions to be drawn, the evidence presented here undoubtedly shows that Phase C has caused a significant deterioration of the Aliakmon River delta surface water quality and a considerable reduction of water discharge.

5.2 *Effects on Sedimentary Regime*

The primary impact of damming on deltaic regions is the reduction of sediment load to the river mouth and coastline erosion. During Phases A, B and C, the Aliakmon River delta underwent major hydrological (Sect. 5.1) and sedimentological changes. Phase A resulted in the reduction of channel length by 20 km. The addition of the Peripheral Canal basin, together with channel straightening and the construction of artificial levee, increased sediment transport to the delta. The Peripheral Canal natural sedimentation rates are estimated between 2.32 and 3.14 $\times$ 10^6 t/y [10], while additional in-channel load due to increased sediment transport capacity of the straightened (higher flow velocities) and steeper (higher hydraulic gradient) channel is expected to have reached the Aliakmon River mouth during Phases A and B. As a consequence, delta progradation proceeded with faster (60 m/y for channel levees) than normal rates [26] and led to the development of a new bird-foot delta (Fig. 5a, b). Average annual

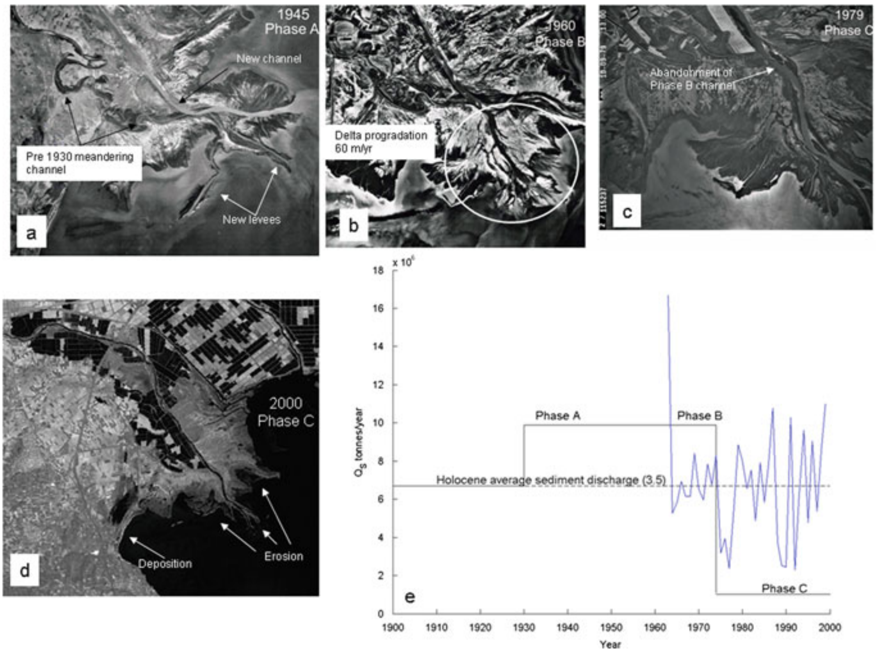


Fig. 5 General review of human-induced changes on Aliakmon River Holocene delta's morphology and sedimentary regime during Phases A, B and C. (**a**) 1945 aerial photo. (**b**) 1960 aerial photo. (**c**) 1979 aerial photo. (**d**) 2000 LANDSAT satellite image. (**e**) Natural (*blue line*) versus human-induced (*black line*) average sediment discharge values at Aliakmon River mouth

sediment discharge at the mouth of the Aliakmon River, between 1916 and 1956 (Phases A and B), has been estimated [16] at 9.9×10^6 t/y (Fig. 5e), a value in agreement with the sum of estimated natural rates of the Aliakmon River (Sect. 3.5) and Peripheral Canal.

Furthermore, the Peripheral Canal drainage basin contribution has altered the mineral composition of deltaic deposits with an observed enrichment in heavy minerals. Measured values at the river mouth indicate high concentrations of Mn, Ni, Co and Cr [27], most likely derived from the weathering of lateritic deposits within the Peripheral Canal basin (northeastern Vermion) [28]. In contrast, mineral compositions of deltaic deposits from sediment cores provided no evidence of heavy mineral transport from the upper Aliakmon River (3.5). Deltaic sediments are characterised by low concentrations of Cu, Pb and Zn [7], a phenomenon linked to the lack of heavy industrial activities along its lower course.

Damming during Phase C caused a 90% reduction of deltaic sedimentation (estimated load 1.03×10^6 t/y), compared to Phase B. More recent (1995–2000) estimates of sediment discharge of the Aliakmon River mouth suggest even lower values of 0.1×10^6 t/y [29]. Consequently, human-induced changes on the sedimentary regime had an immediate impact on the shoreline evolution of the Aliakmon delta, with erosion of the northern part and subsequent siltation of the Methoni Bay (Fig. 5d).

In addition to human-induced alterations of the Aliakmon River sediment transport regime, Phase C caused a pronounced change of channel and deltaic sediment grain-size distributions. Enrichment of deltaic deposits with finer sediments resulted from the reduction of coarse (fine gravel and sand) sediment availability and transport capacity (low discharge).

6 Conclusions

The natural evolution of the Aliakmon River delta has been affected by human activities since the Palaeolithic. During the past 90 years, there has been an intensification of anthropogenic pressures on the river's watershed and delta along three major phases. The former Phases A (1925–1934) and B (1934–1974) of human impacts included a series of hydraulic works that aimed at flood interception, the drainage of swamps and low-lying areas and the construction of an extensive irrigation network. These works were necessary, initially for the hygiene of the newly located immigrant populations and, to a further extent, for the economic development of the area through initiation of systematic agriculture. During these phases, the Aliakmon River delta experienced a 50% sediment transport increase in relation to Holocene pre-anthropogenic rates, an enrichment of deltaic deposits with heavy minerals and higher than normal delta progradation rates. Water quality during this period was 'excellent' as population along the entire watershed was significantly lower, agriculture was confined in smaller than present areas along the delta and the area was lacking industrial units; a few units contributed to limited sources of surface water pollution.

Phase C marked the beginning of a heavily impacted period for the Aliakmon River delta. Damming along the rivers' upper course, population growth along its lower part, use of fertilisers and pesticides for agricultural activities and establishment of food and fruit cannery industrial units, coupled with the ever-increasing demand for electrical power and water, have resulted in a human-controlled system characterised by intense environmental pressures.

More specifically, during Phase C, the Aliakmon River delta experienced a reduction of water discharge towards the river mouth, an alteration of its hydrological regime, a deterioration of its water quality characteristics and a 90% reduction of sediment transport rates compared to Phase B. Reduced water discharge and the interception of sediment transport from the river's upper reaches have resulted to enrichment of river channel and delta front deposits with finer sediments. Fine sediments trap pollutants, enhancing river and seabed pollution, and are easily eroded by waves and longshore currents. As a result, the Aliakmon River delta is currently undergoing erosion and the shoreline is retreating (Fig. 5d). Immobilised material from the Aliakmon River mouth and delta through the prevailing cyclonic circulation is deposited in the Methoni Bay and other locations south of the river mouth, enhancing siltation and degradation of proximal to delta beaches, on which major touristic units have developed during Phase C. Also, the ever-increasing need for water, for urban, industrial and agricultural use, along with increasing pollutants and decreased water discharge in the vicinity of the Aliakmon River, has left very little space for proper regulation of the ecological quality of lower Aliakmon and the deltaic region. Water abstraction poses as a major threat to coastal aquifers, through salinisation.

The future evolution of the entire Aliakmon River fluvial system is not restricted to its lower course and delta but is of great importance to the broader region of Northern Greece. More than two million people directly involved on primary and secondary production and touristic sectors depend in many ways on the Aliakmon River water and sediment resources. Various scenarios of future climate change have demonstrated the vulnerability of the Aliakmon River human-controlled system to even minor hydrological changes. Currently, as Greece is making significant efforts to overcome the ongoing economic crisis, there appears a unique chance for the country to look back to its own natural resources in a financially realistic and environmentally sustainable way. By taking into account the findings of this review, the call for a new realistic strategic plan concerning the future evolution of the Aliakmon River delta, based on a long-term monitoring study, should be put into effect to prevent the continuous degradation of this fragile and otherwise protected (RAMSAR, NATURA 2000) area and to open a new chapter in the economy of Northern Greece.

References

1. NEDECO (1970) Regional development project of the Saloniki Plain, vol A–C . GRONTMY, The Hague. Technical report
2. Syrides G (1990) Lithostratigraphic, biostratigraphic and paleogeographic study of neogene-quaternary sedimentary deposits of Chalkidiki peninsular, Macedonia, Greece. PhD dissertation, University of Thessaloniki (in Greek)
3. Nance RD (2010) Neogene-recent extension on the eastern flank of Mount Olympus, Greece. Tectonophysics 488:282–292
4. Armijo R, Meyer B, Hubert A, Barka A (1999) Westward propagation of north Anatolian fault into the northern Aegean: timing and kinematics. Geology 27:267–270
5. Karageorgis A, Anagnostou C (2003) Seasonal variation in the distribution of suspended particulate matter in the northwest Aegean Sea. J Geophys Res. doi:10.1029/2002JC001672
6. Hyder P, Simpson JH, Christopoulos S, Krestenitis Y (2002) The seasonal cycles of stratification and circulation in the Thermaikos Gulf region of freshwater influence (ROFI) northwest Aegean. Cont Shelf Res 22:2573–2597
7. Poulos SE, Chronis GT, Collins MB, Lykousis V (2000) Thermaikos Gulf coastal system NW Aegean Sea: an overview of water/sediment fluxes in relation to air-land-ocean interactions and human activities. J Mar Syst 25:47–76
8. Poulos SE, Chronis G (1997) The importance of the river systems in the evolution of the Greek coastline. CIESM Sci Ser 3:75–96
9. Skoulikidis N (1993) Significance evaluation of factors controlling river water composition. Environ Geol 22:178–185
10. ECOS Consultants (2001) Water resource management plan of Thessaloniki broader region. Hydrological analysis of Aliakmon River. Greek Ministry of Environment and Public Works, Athens (in Greek)
11. Poulos SE, Collins M, Evans G (1996) Water-sediment fluxes of Greek rivers, southeast alpine Europe: annual yields, seasonal variability delta formation and human impact. Z Geomorphol 40:243–261
12. Konstantinidis KA (1989) Land reclamation project of the plain of Thessaloniki. Geotechnical Chamber of Greece, Thessaloniki (in Greek)
13. Karamanos H, de Madron D, Polyzonis V, Ludvig W (2000) The adjacent land areas as fresh water and particulate matter contributor to the coastal zone of Mediterranean Sea: Gulf of Lions and Thermaikos Gulf. METRO-MED project technical report, Athens, pp 10–19
14. Skoulikidis N, Gritzalis K, Kouvards TH (2002) Hydrochemical and ecological assessment of a Mediterranean river system. Glob Nest Int J 4:29–40
15. Styllas M (2014) A simple approach for defining sequence stratigraphy using borehole and cone penetration test data. Sedimentology 61:444–460
16. Kapsimalis V, Poulos S, Karageorgis A, Pavlakis P, Collins M (2005) Recent evolution of a Mediterranean deltaic coastal zone: human impacts on the inner Thermaikos Gulf, NW Aegean Sea. J Geol Soc 162:897–908
17. Lykoussis V, Karageorgis A, Chronis G (2005) Delta progradation and sediment fluxes since the last glacial in Thermaikos Gulf and Sporades basin, NW Aegean Sea, Greece. Mar Geol 222–223:381–397
18. Fouache E, Ghilardi M, Vouvalidis K, Syrides G, Kunesch S, Styllas M, Stiros S (2008) Contribution on the Holocene reconstruction of Thessaloniki plain north-central Greece. J Coast Res 25:1161–1173
19. Ghilardi M, Psomiadis M, Cordier S, Delanghe-Sabatier D, Demory F, Hamidi F, Paraschou T, Dotsika E, Fouache E (2011) The impact of rapid early- to mid- Holocene paleoenvironmental changes on Neolithic settlement at Nea Nikomedia, Thessaloniki Plain, Greece. Quatern Int. doi:10.1016/j.quaint.2010.12.016
20. Bintliff J (1976) The plain of western Macedonia and the Neolithic site of Nea Nikomedia. Proc Prehist Soc 42:241–262

21. Zalidis G, Dimitriadis X, Antonopoulos A, Gerakis A (1997) Estimation of network irrigation efficiency to cope with reduced water supply. Irrig Drain Syst 11:337–345
22. Skoulikidis N, Amaxidis Y (2009) Origin and dynamics of dissolved and particulate nutrients in a minimally disturbed Mediterranean river with intermittent flow. J Hydrol 372:218–229
23. Nikolaidis NP, Karageorgis AP, Kapsimalis V, Marconis G, Drakopoulou P, Kontogiannis H, Krasakopoulou E, Pavlidou K, Pagou A (2006) Circulation and nutrient modelling of Thermaikos Gulf, Greece. J Mar Syst 60:51–62
24. Lazaridou-Dimitriadou M, Artemiadou V, Yfantis G, Mourelatos S, Mylopoulos Y (2000) Contribution to the ecological quality of Aliakmon river (Macedonia, Greece): a multivariate approach. Hydrobiologia 410:47–58
25. Ilias FI, Lakis C, Papazafeiriou AZ (2008) Environmental parameters affecting species diversity along the Aliakmon River, north Greece. Pak J Biol Sci 11:862–868
26. Kakogiannou E, Soliopoulou A, Feneri AM (2012) The use of temporal remote sensing images and GIS for Aliakmon delta coastline detection. South East Eur J Earth Obs Geomat 1:55–61
27. Chester R, Voutsinou FG (1981) The initial assessment of trace metal pollution in coastal sediments. Mar Pollut Bull 12:84–91
28. Economou-Eliopoulos M (2002) Apatite and Mn, Zn, Co-enriched chromite in Ni-laterites of northern Greece and their genetic significance. J Geochem Explore 80:41–54
29. Karageorgis AP, Anagnostou C (2001) Particulate matter spatial-temporal distribution and associated surface sediment properties: Thermaikos Gulf and Sporades Basin, NW Aegean Sea. Cont Shelf Res 21:2141–2153

The Biogeographic Characteristics of the River Basins of Greece

Stamatis Zogaris and Alcibiades N. Economou

Abstract Biogeographic regionalizations provide frameworks for a holistic understanding of river basin areas and their inland water ecosystems. Here we employ the freshwater ecoregion concept to outline biogeographic aspects of the aquatic and semiaquatic biota and river ecosystems in Greece. Emphasis is given to freshwater fishes which cannot readily disperse over mountain watershed barriers and marine areas; they are utilized as primary biogeographic indicators. Although various biogeographic regionalization maps are surveyed, the Freshwater Ecoregions of the World (FEOW) initiative is adopted, and this review helps redefine certain recently published ecoregional boundaries in Greece. Along with freshwater fishes, other animal and plant distributions and knowledge of geological history and climatic patterns help guide the boundary definition of eight freshwater ecoregions in Greece. Gaps in knowledge concerning species distributions and taxonomy as well as the biogeographic understanding of each freshwater ecoregion are assessed.

Keywords Aquatic biota, Conservation, Freshwater biodiversity, Freshwater ecoregions, Freshwater fishes

Contents

S. Zogaris (✉) and A.N. Economou
Institute of Marine Biological Resources and Inland Waters, Hellenic Centre for Marine Research, 46.7 km Athens-Sounio Av., 19013 Anavissos, Greece
e-mail: zogaris@hcmr.gr

N. Skoulikidis et al. (eds.), *The Rivers of Greece: Evolution, Current Status and Perspectives*, Hdb Env Chem (2018) 59: 53–96, DOI 10.1007/698_2017_475,

1 Introduction

River basins are virtually "biogeographic islands" for freshwater biota. At a regional spatial scale, large areas with aquatic ecosystems that share species assemblages can be described as "ecosystem regions" or ecoregions. Robert Bailey [1], who championed in ecoregional cartography, defined ecoregions as "major ecosystems" ... "resulting from large-scale predictable patterns of solar radiation and moisture that, in turn, affect the distribution of local ecosystems and their component plant and animal species." For the realm of inland water ecosystems, these regional entities are appropriately termed freshwater ecoregions [2], and they have become key geographical units for aquatic ecosystem inventories, monitoring, and conservation planning in recent years [3]. Ecoregions are also important conceptual frameworks to describe and evaluate a country's biodiversity and natural aquatic resources.

Ecoregional classifications have been widely used as a first-tier screening in procedures for classifying water resources, and they are also a key geographical criterion for river typologies within Europe's Water Framework Directive – WFD 2000/60/EC [4]. Many researchers have called for a hierarchical river classification, where a regional or ecoregional typological criterion tops the standardized classification framework (e.g., [5, 6]) (Fig. 1). In 2008, a global assessment to delineate the Earth's freshwater ecoregions was promoted by conservation organizations [8, 9].

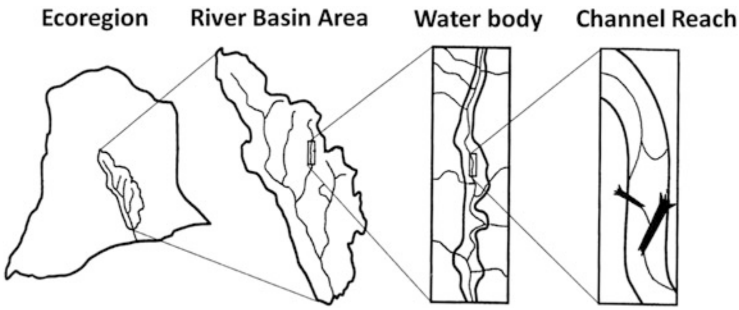


Fig. 1 Scales of river classification that are now part of policy-relevant river management, conservation planning, and monitoring; ecoregions or other biogeographical regions top the hierarchy (adapted from [6] as presented in [7])

This project produced practical classification and regional delineation criteria and a first baseline charting freshwater ecoregions on a global scale.

The Freshwater Ecoregions of the World (FEOW) project is primarily a freshwater biogeography scheme. It is guided by the influences of freshwater species' phylogenetic history, paleogeography, and ecosystem distribution patterns in order to delineate regions exhibiting relative homogeneity of aquatic ecosystem structure [8]. This regionalization mainly utilized freshwater fish species distributions as proxies for the distinctiveness of wholly aquatic biotic assemblages [3]. FEOW utilized watershed lines and deep marine waters as criteria for boundaries, unlike the older "terrestrial" ecoregional schemes which give overriding value to potential natural vegetation and general physiographic and climatic characteristics [10–12]. Expert judgment was important in making final boundary decisions and a prestigious panel of biodiversity experts contributed to the freshwater ecoregion delineations [9].

Regionalization schemes such as ecoregion mapping show varying scales of regional analysis. The freshwater ecoregional scale is definitely spatially restricted relative to the much broader "biome" scale or the recently redefined "major zoogeographical region" scale [13]. For the smallest freshwater ecoregional units, it is difficult to define a size limit; they usually extend for several hundreds of kilometers and often include at least several dozens of more or less biotically similar river basin areas (i.e., a simple rule of thumb being that "each basin is a state, each ecoregion a country"). However, in exceptional cases there are some rather small freshwater ecoregions, such as some very large tropical lake systems and large islands which are characterized by millions of years of isolation and outstanding evolutionary divergence [2]. Greece, for example, is a country on a "biogeographic crossroads" where rather small ecoregions meet. The territory of Greece includes eight freshwater ecoregional units, the largest number of ecoregions of any EU country (Fig. 2).

In this chapter, we explore the biogeographic ecoregions that encompass Greece's river basin areas, and we survey and interpret the FEOW ecoregional delineations. Controversy and disagreements on boundaries persist, and the issue of completing an accurate ecoregional map for the freshwater realm is still in progress. After many

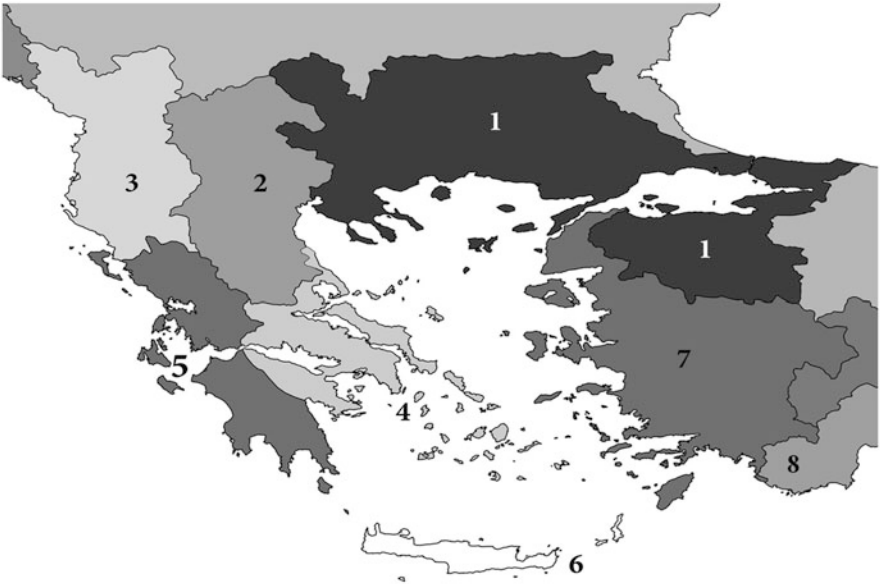


Fig. 2 The freshwater ecoregion delineations of Greece and the surrounding lands as developed by the Conservation Science Program of World Wildlife Fund and The Nature Conservancy [9]. There are still disagreements among some of the boundaries; see text (*numbers* have been superimposed and correspond to Table 1). Map extracted from FEOW [9]

years of working with the ecoregional framework at the Hellenic Centre for Marine Research (HCMR), we propose specific boundary changes to the FEOW map and we document them, but this is not the place to do this exhaustively. Although we use the relevant biogeographic literature and previous fish-based biogeographic analyses, we do not dwell on describing the historical biogeography in detail (i.e., interpreting biogeographic dispersal episodes or routes or the genesis of current biotic assemblages). The ultimate goal here is an introductory regional-scale review of freshwater lotic ecosystems and their biota.

2 Previous Biogeographic Delineations

Due to its remarkable position among three continents and its diverse and fragmented mountain chains and archipelagos, Greece has been a focus area for biogeographic research (e.g., [16, 17]). However, the overwhelming majority of research has focused on terrestrial biogeography; terrestrial species of plants and animals on the islands have dominated biogeographic work for long periods [18–20]. The Aegean is one of the world's hotspots for island biogeographic research, and many new theoretical and analytical approaches and interpretations have been produced in this area [21–24]. This biogeographic research has also helped to integrate a huge body of literature from various research endeavors, including geology, paleontology, archeology, climatology, botany, zoology, ecology, and conservation science. Biogeography is particularly important for exploring systematics and taxonomy, especially in areas where species have evolved in isolation or retained relic populations. As a result of decades of biogeographic research, many disparate biogeographic maps have been published, using very different indicator taxa groups (e.g., terrestrial invertebrates, reptiles, the paleofauna, endemic terrestrial flora, potential natural vegetation, etc.).

Botanists have contributed significantly to biogeographic regionalizations in Greece. Turrill [19] in 1929 was the first to divide the country into six phytogeographical regions. Later, Rechinger (1943) [18] first addressed the phytogeographical peculiarities of the Aegean and discovered the important biogeographic boundary along the mid-Aegean trench, known now as "Rechinger's line" – the biogeographic divide between the European and Asian Aegean [17]. Ganiatsas [25] also produced a phytogeographical map based largely on Rechinger's work (Fig. 3a). Some years later, Strid [17] divided Greece into 13 phytogeographical regions, and this practical compartmentalization, also based on the previous phytogeographical regionalizations, has remained unchallenged and is widely used today [27, 28].

The zoologists also charted biogeographic boundaries in Greece. Often these "zoogeographical maps" were exclusively for specific taxonomic groups, including on some occasions freshwater aquatic and semiaquatic groups (Fig. 3b, c). Distributional knowledge of the amazing array of arthropod diversity in Greek aquatic ecosystems is limited in part because of the lack of local taxonomic specialists and the relatively late beginning of taxonomic studies. It should be pointed out that by the late 1970s and early 1980s, zoogeographic interest in Greece helped establish the International Congress on Zoogeography of Greece and Adjacent Regions (ICZEGAR) first promoted by Prof. I. Matsakis and by many "philhellene" European biologists who had been collecting biological material throughout Greece. ICZEGAR is still a hive of development for biogeography in Greece and the wider region [29].

Regarding aquatic animals, freshwater fishes dominate biogeographic research in Greece. We owe a lot of baseline work to Prof. PS Economidis, an

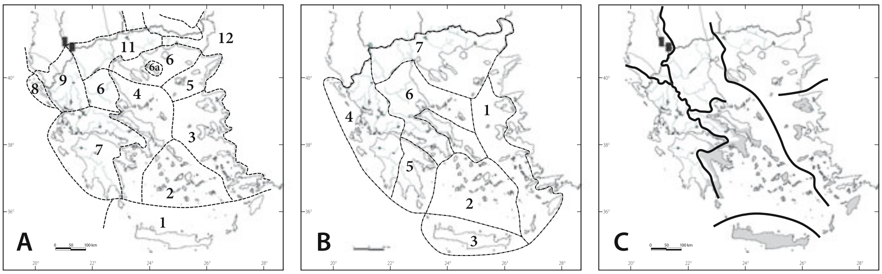


Fig. 3 Important biogeographical delineations for various biotas: (**a**) Plants [25]; (**b**) *Hydraenidae* beetles [26], and (**c**) major freshwater biota assemblage breakpoints (key barriers to dispersal) mainly using fish distributions (this study); *gray-colored areas* in map (**c**) show areas where intermittent and ephemeral stream flows dominate lotic waters, due to seasonally semiarid conditions and geology (adapted from [7])

intrepid explorer who collected and collaborated with many ichthyologists since the late 1960s (e.g., [30, 31], and references therein). However, most modern fish-based biogeographic maps are either rather repetitive or their boundaries vary with respect to changing species taxonomies (nomenclatural changes) and species subsets utilized in the analyses (e.g. [32–34]). Different spatial scales in the regionalizations also influence biogeographic boundaries; broadscale applications necessarily create unconventionally much larger and fewer regional units (e.g., the Europe-wide analysis of Reyjol et al. [35]). In recent ichthyogeographic delineations, only the most prominent boundaries resulting from the river basin fish assemblage classification analyses are charted, and specific criteria for boundary setting are set [36, 37], while more sophisticated quantitative methods are applied (e.g., [38, 39]).

Compared to fishes, there is limited work on other freshwater biota in Greece or the neighboring Balkan and Asian countries. At the continental scale, with respect to the European freshwater biota, spatially broad and rather crude freshwater biogeographic regional delineations were originally proposed by J. Illies [40] in his multi-taxa freshwater zoogeographical compilation *Limnofauna Europaea*. However, Illies's freshwater ecoregions have been seriously criticized and are now considered outdated and partially flawed [36]. Despite this dispute, Illies' boundaries were used for partitioning European ecoregions for Europe's Water Framework Directive [4] inland water typologies. One reason for the difficulty of producing freshwater ecoregional maps has been the many varied and disparate distributional patterns of very different "freshwater" biota; some taxa are wholly aquatic, large-sized poor dispersers (such as mussels), while many are semiaquatic or can even disperse terrestrially by flying across biogeographic barriers (i.e., many semiaquatic insects). A major problem with aquatic invertebrate work is that inventories and distributional surveys for many species are incomplete or poorly surveyed [41, 42].

Biogeographic ecoregion regionalizations have thus encountered controversy; different spatial scales, statistical methods, and methodological delineation procedures are used, and many biogeographic maps of the same area give quite different boundaries. For example, Illies's [40] biogeographic maps based on many animal taxa were altered with different published editions and later politically relevant usage within the WFD, i.e., using state borders instead of biogeographic characteristics (Fig. 4).

Aquatic biogeographic cartographers look for common biotic breaks in species distributions in order to chart regional boundaries. For example, the concept of faunal break boundaries refers to specific boundary lines of rapid faunal change that are usually associated with prominent long-standing geographical, geological, marine, and/or climatic barriers to species dispersal, such as watershed lines; see Fig. 3c. Faunal break boundaries are obviously scale dependent, and the degree of dissimilarity will vary based on taxonomic groups used, their dispersal abilities, and several other parameters. Misinterpretations or differing opinions easily get published [36].

Fig. 4 Gradual changes in the broadscale freshwater zoogeographic "ecoregions" of Illies's *Limnofauna Europaea* (the first two maps from the *left*, with published volume dates) and the final ecoregion map used in the Water Framework Directive [4]. For ecoregion names, *numbers* are provided in Table 1 (figure redrawn from [7])

Finally, we should mention that regional syntheses based on terrestrial ecoregional concepts are probably even more difficult to standardize than the freshwater biogeographic approaches, since even more parameters are introduced to the "ecoregional" perspective [10, 43]. Since distributional patterns of many freshwater taxa and the ecosystem processes that sustain them do not usually correspond well to terrestrial ecoregion boundaries, we endorse the development of separate freshwater and terrestrial frameworks for conservation-based analyses [2].

Generally, modern terrestrial ecoregional maps vary among scientific teams and subdisciplines, and this has sometimes created confusion [44]. A recent conservation-relevant procedure focused on terrestrial ecoregions of the world [11, 45] using potential natural vegetation categories and various species distributions among other criteria (Fig. 5). The authors of this map do warn that "no single biogeographic framework is optimal for all taxa and ecoregions reflect the best compromise for as many taxa as possible" [11]; boundaries rarely form abrupt edges but are bound by ecotones and mosaics. Our opinion is that this work is a gross generalization and does not compare well with the diversity of other potential natural vegetation renditions in Southeast Europe [46] or even traditional biogeoclimatic cartography [47] for the region. Olson et al.'s [11] global terrestrial ecoregional map was significantly revised in some parts of the Earth's surface (i.e., Arabian Peninsula) in its reissue in 2017 [45], but the ecoregional delineations in the Balkans remain unchanged. Nonetheless, the conservation-relevant gap analyses using this map have produced a very useful global conservation evaluation [45].

3 Geological Setting

Greece exhibits a unique geophysical diversity and a tumultuous geological history in its 132,000 km^2 area. It has over 3,000 islands, and if we include islets and rock stacks, the number surpasses 7,800 islands and islets [48]. Greece is also a land of

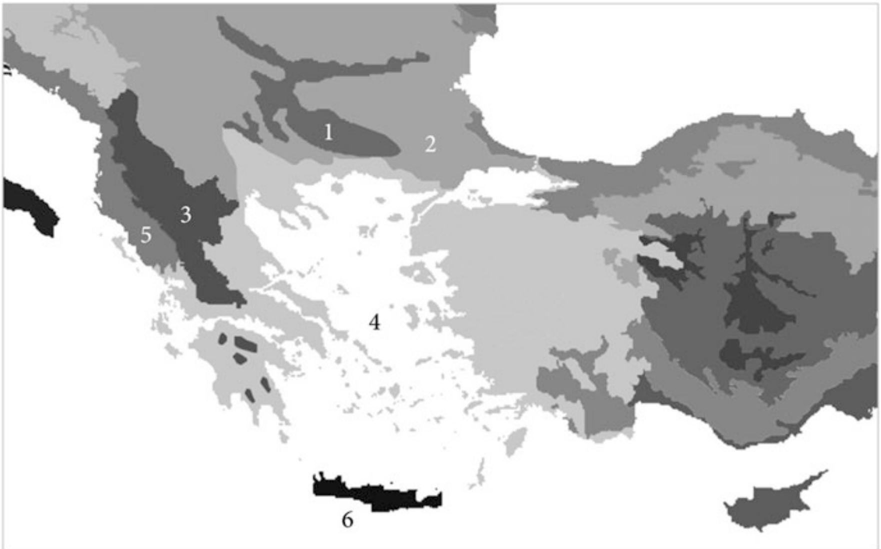


Fig. 5 Segment of the terrestrial ecoregions of the world map by Dinerstein et al. [45]. Six ecoregions are delineated in Greece's territory: (1) Rhodope montane mixed forests, (2) Balkan mixed forests, (3) Pindus mountains mixed forests, (4) Aegean and Western Turkey sclerophyllous and mixed forests, (5) Illyrian deciduous forests, and (6) Crete Mediterranean forests. Extracted map adapted from Dinerstein et al. [45]

hundreds of peninsulas. Although only about 20% of the land area is made up of islands and islets, there is a uniquely convoluted and incredibly long coastline with hundreds of autonomous river basin areas. An overwhelming number of river basins are very small; many small non-perennial streams and torrents dominate in the islands, peninsulas, and dry coasts. In contrast, some sizable rivers, including large transboundary basins, exist in the northwest and north. This globally unique geographical configuration is a result of the tectonically active geology of the southern Balkans and its surrounding regions [49, 50].

The geological history of the Balkans and the Aegean region is complex, involving dynamic tectonic action and long periods of orogenesis that created an evolving geographical scene effecting the distribution and diversity of the freshwater biota. As outlined by Bănărescu [14] and Skoulikidis et al. [51], the following main attributes seem to be the major geological events that produced such complex biogeographic patterns in the freshwater biota of the southern Balkans:

1. The existence of the former Tethys Sea (in the region between the current Mediterranean-Danube Valley-Black Sea-Caspian Sea).
2. The orogenic upheaval of the Carpatho-Balkans separating the wider Tethys into a southern (Mediterranean) and northern sector, the Paratethys, much of which later formed large lakes or dried out.

3. The orogenic upheaval of the long and narrow Pindus cordillera (an extension of the Dinaric Alps) separating Greece and the Western Balkans into east-west biogeographic sectors.
4. The existence of many ancient lakes, many derived from various parts of the Tethys and Paratethys, which host many long-isolated endemic species.
5. Fluctuations of sea levels in the Adriatic enabling faunal exchange between Italy, the Northern Adriatic and the Western Balkans.
6. The continental contact and separation between the southern part of the Balkans and Anatolia: until the beginning of the Middle Miocene, the southern Balkans and Asia Minor comprised a continuous composite landmass (Fig. 6a); this was interrupted by the Aegean landmass subsidence during the late Miocene (Fig. 6b).
7. The existence of archipelagos due to remarkable tectonic diversity and volcanism in the Aegean, including the creation of a distinct southern island mass (Crete) (Fig. 6c).
8. The desiccation of the Mediterranean at several periods, but especially during the Messinian Salinity Crisis (from 5.96 to 5.33 MYA), which favored freshwater flows and connections among formerly disjunct river basins.
9. The Pleistocene glaciations which dropped eustatic sea levels down to approximately 120 m and created connections among many river basins (Fig. 6d) but also created climatically benign "refugia" in the southern coastal and lowland parts of our area.
10. The gradual modification of river networks through repeated river captures (river piracy in tectonically active areas) which blended the headwaters of rivers and helped biota dispersal over previously impenetrable mountain watersheds.

4 Climate

The Mediterranean climate was established about 3.2 MYA [53], and during the Holocene, most of the territories belonging to Greece have been typically Mediterranean in climate. The Mediterranean climate is diverse, and there are important variants that affect the water cycle, hydrological flow regimes in surface waters, and the aquatic communities. Precipitation is irregularly distributed across Greece, ranging from 1,300 mm annually in northwestern Greece (especially in the Tzoumerka mountains) to 300 mm in the southeast coastal rain shadow areas [54]. The west side of Greece, west of the Pindus mountain range, is much wetter and more humid than the eastern part. In much of eastern continental Greece, rain shadow areas create pockets of seasonally arid conditions with high evapotranspiration rates and a long summer drought. These rain shadows create conditions where non-perennial river systems dominate, since long-term droughts define the character of flow regimes [55]. Hydrology and natural flow regimes vary remarkably among basins and longitudinally along the river courses, depending on local climate and the geological,

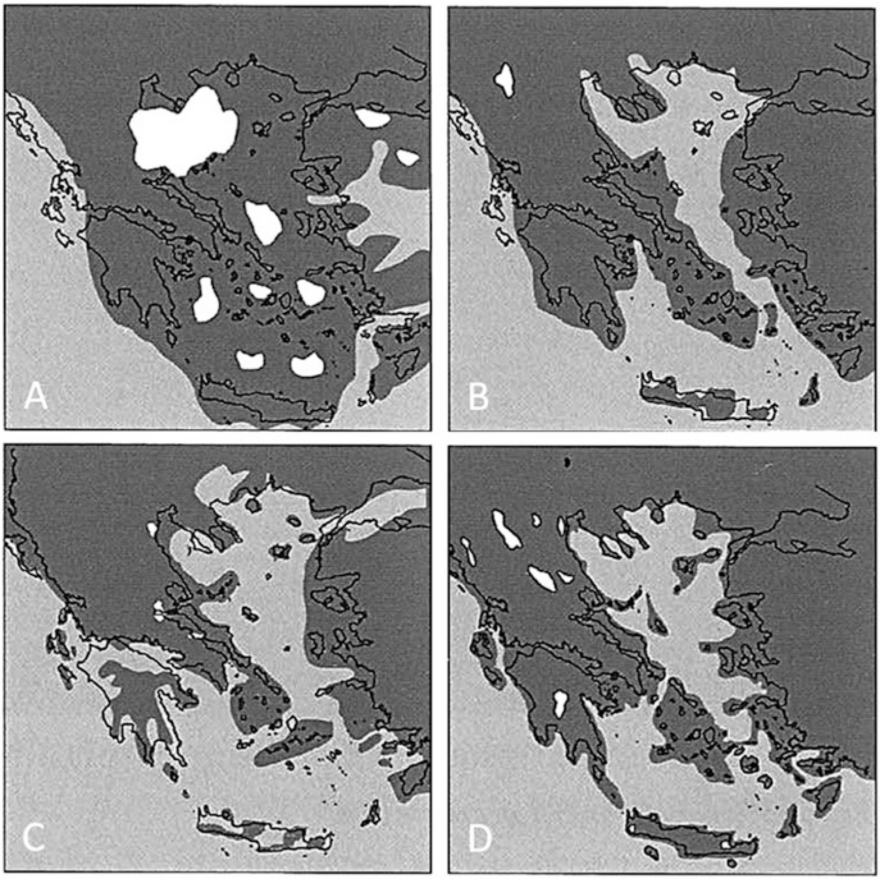


Fig. 6 Examples of paleogeographic change in Greece: landmasses in *dark gray*, lakes in *white* (the present geography outlined within the images). (**a**) Middle Miocene 12 MYA, with the Aegean being a continual landmass; (**b**) late Miocene 8 MYA, first breakup of the Aegean; (**c**) Pliocene 3.5 MYA; south Aegean arc islands including Cretan paleo-islands. (**d**) Middle Pleistocene 0.8 MYA; lowering eustatic water levels during glacials reconnect islands, and many river basins merge. Selected map depictions originally from Valakos et al. [52] and modified in Zogaris [7]

geomorphological, and topographical conditions. Due to the predominance of calcareous geology in western and southern Greece, karstic springs are very important in providing steady flow regimes to relatively small rivers [51]. Spring-fed streams are therefore quite incongruous "oases" and are often associated with geological patterns and a steep mountain relief. In contrast, disturbance regimes such as flash floods and severe droughts are also common, often even in small seasonally arid torrents. Climatic research has shown that important aspects that affect lotic waters and their biota in the Mediterranean are droughts, and a remarkable variability of precipitation pattern has been documented through the centuries [56]. It is certain that even before the high-intensity water exploitation of modern times, there have been "epic megadroughts" that may have had an important role to play in the locally impoverished state of insular and peninsular aquatic faunas and floras. Finally, the influence of anthropogenic water abstraction and climatic variability or recent climatic change create mixed stresses on running water ecosystems, and it is often difficult to interpret what is modified solely by human activities [57].

5 The Aquatic and Riparian Biota

The southern Balkans and Anatolia are famous as acting as biological refugia during the past glacial periods, thus conserving many Eurasian aquatic and semiaquatic species that were extirpated by climate change in much of temperate Europe. Isolation and vicariance (i.e., the splitting of populations by barriers to dispersal) have created conditions for the evolution of a large percentage of range-restricted species, known as local endemics. Endemics can be defined to a geographical entity (such as a region or the state's political borders). One of the most outstanding attributes of using biogeographic indicators is the identification of areas where these range-restricted species concentrate. The territory of Greece has the highest percentage of endemic fishes in the EU [58] and may equally be important as a center of endemism for aquatic, semiaquatic invertebrates and aquatic species parasites as well [59]. In contrast, many aquatic microorganisms are generally very widespread (cyanobacteria, fungi, and microscopic plants) since they are dispersed by atmospheric phenomena or larger animals [41]. Evidence of the cosmopolitan distribution of lentic planktonic organisms and other microscopic plants and animals has been documented in Greece [38]. Biogeographic patterns can certainly be gathered for interpretation purposes from reviewing a wide range of aquatic and semiaquatic species distributions. Below we provide a summary of some well-known aquatic species groups with notes on their usefulness as biogeographic indicators (Fig. 7).

5.1 Aquatic and Riparian Plants

Plant distributions and vegetation formations are informative for ecoregional cartography [10]. In Greece, most botanists have focused on terrestrial species, terrestrial vegetation, and particularly on the floral distribution patterns in the archipelagos and mountains [60]. Aquatic plants were less interesting for early plant biologists and collectors since most of the range-restricted species are actually dry-land species, often inhabiting rocky outcrops, mountain cliffs, etc. In fact, Greece is a global center of plant endemism for many such species groups: the country sustains 1,462 endemic plant taxa (species and subspecies) which roughly amounts to 22% of its 6,600 documented

Fig. 7 Various animal taxa that provide biogeographic knowledge. (**a**) The Freshwater crab *Potamon pelops* from the Peloponnese (Alpheios river); (**b**) unidentified unionid mussels from the Western Aegean Ecoregion (Spercheios river); (**c**) endemic *Rhodeus meridionalis* from the Macedonia-Thessaly Ecoregion; (**d**) endemic Greek stickleback *Pungitius hellenicus* and a water snail (*Theodoxus* sp.) from the Western Aegean Ecoregion; (**e**) terrestrial salamander, *Lyciasalamandra luschani* from the Southern Anatolian Ecoregion (Kastellorizo island). (Photos (**a**) I. Strachinis; (**b**) A. Christopoulos; (**c**, **d**) S. Zogaris; (**e**) K. Sotiropoulos)

taxa [27]. Most of the local endemics are in the southern part of Greece, the islands and mountains [61, 62].

Unlike the "taxonomically attractive" local endemics, the aquatic and "water-loving" (hydrophilous) plant species are geographically rather widespread. Many hydrophilous plants are easily dispersed primarily by atmospheric phenomena, migratory birds, and other animals. Recent reviews show a poor documentation of hydrophilous species (e.g., [27, 63, 64]). Some species distributions are poorly known because their aquatic and wetland habitats have suffered much change in Greece during the last century, and the smaller wetland habitats have been poorly surveyed [65, 66].

Since hydrophilous plants are particularly dependent on specific aquatic and humid habitat conditions, some species that are "intolerant" to drought (or the effects of seasonal water scarcity) have become locally or regionally extirpated. Despite their widely scattered distributions, some species groups, even such as the charophytes (macroalgae in the family *Characeae*), include taxa that are said to have localized distributions, and some are considered threatened [63]. Species that need deeper waters and stable lentic conditions in rivers, such as water lilies and water chestnuts, for example, are scarce in the southern half of the Hellenic peninsula and its islands; these are locally common in the larger river basins of the north and northwest of the country. In fact, one of the few recent plant extinctions in Greece includes a water plant, *Stratiotes aloides* (Hydrocharitaceae), which was used to be found in north-central Greece [27]. It is unusual that only a very few hydrophilous plants are featured in the Greek Red Data Book account of 2009 [28]. The Red Data Book does, however, include some interesting and rare Mediterranean endemic species such as *Callitriche pulchra* (Gavdos, Crete ecoregion) and the riparian orchid *Dactylorhiza pythagorae* (Samos, Eastern Aegean Ecoregion), but information of many important species and groups such as the charophytes is scant and poorly inventoried and documented [63, 66]. Finally, the bibliography on river plants is very limited for such a rich country in streams and the value of these plants as bioindicators (e.g., [67]).

Riparian vegetation includes all wetland and riverside or lakeside vegetation that is influenced by the adjacent water body. European assemblages are very similar to the water plants in being rather widespread. However, there are several distinct riparian plant communities with biogeographic characteristics defining particular patterns especially in Southern Europe [68] and parts of Greece [69]. Interesting and widespread riparian trees in Greece include the oriental plane (*Platanus orientalis*) and bay laurel (*Laurus nobilis*); these are relics from the Tertiary period that have long become extirpated from other parts of Southern Europe. Some very rare woody plant assemblages exist that have high biodiversity and biogeographic interest. These include river riparian stands of the oriental sweetgum *Liquidambar orientalis* on Rhodes Island and the Cretan date palm *Phoenix theophrasti*, restricted to Crete and Southwestern Anatolia. In Greece, a north-south gradient in species richness in river riparian woody plants can be detected, with northern montane assemblages being much richer in species than the south [70]. This pattern may be also due to anthropogenic degradation since southernmost areas have had much denser human populations along river valleys than the more extensive mountain areas of the north.

5.2 Aquatic Invertebrates

Although not a single taxonomic group, this is definitely the most important animal assemblage in terms of sheer biomass and species diversity in inland waters, wetlands, and riparian zones. The aquatic invertebrates include many groups such as arthropods, flatworms, worms, gastropods, bivalves, and a myriad of microscopic forms that are found in nearly all running water environments, even in intermittent or near-ephemeral stream conditions. Some of these organisms disperse among water bodies via the atmosphere (many insects for example have a flying stage); others use dispersal vectors such as birds [71, 72]. Relatively poor interbasin dispersers are the wholly aquatic benthic groups such as the bivalves [73] and several gastropods [74]. Many spring-inhabiting snails are restricted to specific parts of Greece, and many new endemic species are still being described [75–77]. Some aquatic insects such as the aquatic beetles (Coleoptera) in the family Hydraenidae also include relatively poor dispersers [26, 78], and several other beetle groups also have many range-restricted species, many endemic to parts of southern Greece [79]. Generally, it is estimated that Greece has more than 4,000 endemic invertebrate species, an incredible number for such a small country [80].

Endemicity of macroinvertebrate species seems to be highest in the south and west of the country and in the islands [81], and this is also reflected in many terrestrial invertebrates [79, 82, 83]. Prominent barriers to lowland species are the main mountain chains, especially the Pindus. In some groups such as the planarian flatworms (Genus *Dugesia*, Platyhelminthes), initial patterns of distribution are surprisingly similar to aquatic vertebrates [84]. The larger islands and peninsulas have more endemics; Crete, for example, is an important endemicity hotspot. In the southern part of Greece and the west, even widely distributed insect groups such as the dragonflies (Odonata) and stone flies (Plecoptera) have endemic species [80]. Freshwater crabs are widespread in Greek streams and rivers (species belonging to the genus *Potamon*), some considered endemic to parts of Greece. Of the crustaceans, freshwater shrimps and amphipods are also especially interesting biogeographic indicators as well [85, 86].

Lastly, general patterns also show Greece's high species richness, since many species from nearby biomes and zoogeographic regions enter the territory [87–90]. Greece is located way to the east and south on the European continent and hosts many interesting species that have their centers of origins in Asia, examples being the huge Bellostomid water bug (*Lethocerus patruelis*) which just reaches Europe solely in the extreme southeast and the damselfly *Epallage fatime* which has mostly southwestern Asian distribution.

It is remarkable that the aquatic invertebrates are still so poorly studied in Greece; even work on common and rather large-sized groups is preliminary or at a developmental stage (e.g., [42, 73, 91, 92]). Despite the outstanding importance of the larger benthic macroinvertebrates, for stream typology and ecosystem understanding, few works exist on species assemblages in specific running water ecosystem types. Several works focus on general patterns of distribution at family or genus level (e.g., [93, 94]) mainly for the purposes of ecological status assessment and monitoring. Moreover, many crenobionts – the spring-inhabiting aquatic invertebrates – show remarkable endemism, and many may be threatened with extinction [76, 95]. Unfortunately, the most recent Greek Red Data Book compilation in 2009 pays little attention to many groups of aquatic invertebrate species primarily due to the grave lack of completeness in documented distributional knowledge [80].

5.3 Freshwater Fishes

Freshwater fishes are the most widely utilized biogeographic indicators in inland waters at the regional scale, and they are rather well studied in Greece. There is much to be learned by utilizing fishes in conservation biogeography [96], and their importance as biogeographic indicators has been corroborated by historical biogeographic evidence [15, 34] and new analytical methods [97]. Importantly, fish proved to be good vehicles to explore phylogeography with new and developing genetic molecular methods and analyses [98]. Extensive collections of genetic samples that grace museum and academic archives from nearly all fish populations in our area of study and the surrounding states have been rigorously studied during the last two decades. The genetic work has assisted in making quantum leaps for taxonomic clarification and the understanding of biotic affinities among taxa and their constituent river basin areas [58, 99]. Thus, the overriding importance of fishes in regionalization work has dominated recent freshwater biogeographically based ecoregional delineations [8]. The importance of freshwater fish geographical distributions has created increasing interest in taxonomy, as has the description of many new range-restricted species, especially in endemic-rich areas, such as the Balkans and western Asia. Fishes are one of the earliest biogeographic elements explored in the Balkans [100], and interest in their distribution patterns has continued to produce many and varied fish-based biogeographic maps.

Greece is the richest country in the EU in terms of its endemic fish species and one of the most important for fish conservation in the Mediterranean [101]. At least 47 out of its 160 freshwater fish species are now considered exclusively endemic within the country's boundaries; several more taxa – at least 15 – are near endemic, i.e., confined to Greece and the near-border frontier areas with neighboring Balkan states (i.e., shared water bodies such as Prespa, Doirani, and Butrint basins). Hydrographic isolation and vicariance are the main factors responsible for Greece's ichthyofaunal diversity. Fish-based biogeographers are not only interested in the endemics; they carefully use all species that are more or less intolerant of seawater and do not disperse through the seas. Primary freshwater fishes are those with little or no tolerance of brackish water (i.e., water with more than 0.5 g per liter total dissolved mineral salts), while secondary freshwater species are tolerant of brackish waters but normally occur in inland aquatic systems rather than the sea, and some are capable of occasionally crossing narrow sea barriers [102]. Primary or primary-like freshwater fishes are by nature confined to freshwater island-like basins, and many of these species have been confined for several millions of years [33, 98]. It should be noted, however, that the arithmetic figures of endemism attributed to the country are changing almost every year as range extensions are discovered (e.g., particularly in Albania; see [103, 104]). Furthermore, the actual number of inland water fish species is also not easily determined, as various marine species are often encountered in the lower sections of inland waters, while new translocated species are becoming established in the wild, particularly in lake and reservoir systems. Finally, taxonomic changes continue taking place at a rapid pace, with new species being described, former synonyms being reinstated, and former "subspecies" validated to species rank. This dynamic state of seemingly perpetual taxonomic change may be confusing, but the situation is clearing up as detailed checklists are regularly revised [58, 105].

5.4 Reptiles and Amphibians

Terrestrial reptiles are at the forefront of zoogeographical research in Greece [16, 23]. The country hosts at least 63 reptile and 23 amphibian species, thus being particularly rich in European terms [106]. Semiaquatic amphibians are less studied, but there are very good case studies that inform biogeographic patterns [107]. Endemic amphibian taxa are confined as endemics to Crete, Karpathos, and some other areas [80]. Also as in fishes, the Pindus mountain range acts as a prominent biogeographic barrier; a near-endemic frog and some newt species, and some terrestrial reptiles, are found only along the west coast of Greece and Albania, bounded by the Pindus [52]. As are fishes, many reptiles are liable to extinction in more fragmented isolated geographical areas such as peninsulas and islands [108]. Reptile species taxonomy has seen many recent changes and much informative phylogeographic research at the genetic level, similar to the changes seen in fishes in the first years of the twenty-first century.

5.5 Birds

Birds are well studied in Greece [109]. Although they are the best interregional dispersers and many species undergo mass migrations, there are some species with distinct biogeographic regional distributions. For example, species with an “eastern distribution” are found most frequently or in larger concentrations in northeastern Greece and the larger wetlands of the Eastern Aegean islands [20, 110]. Some characteristic and iconic waterbirds are collectively near absent from the western parts of Greece (i.e., west of the Pindus range). Many species seem to be much more abundant and frequent in northeastern Greece despite the fact that adequate and extensive wetland habitats exist along the coast of western Greece, and this relates to the important land bridge migration flyways of Thrace. The long cordillera of the Pindus mountains is an important boundary that affects bird migration [109], and even certain long-distance dispersers such as pelicans are known to have different subpopulations on either side of the Pindus cordillera [111]. In rivers it is difficult to tally the full number of birds using exclusively aquatic or riparian habitats; many terrestrial species also use the waters [41]. Finally, birds and especially the waterbirds are very important for the dispersal of plants, animals, microbes, and fungi (seeds, cists, spores, etc. attached to the mud on birds’ legs and feathers) [112]. A large suite of aquatic organisms can also survive passage through the digestive systems of birds because of a digestive trade-off in many birds [72]. In Greece, the north-south distribution of many widespread plant and small aquatic invertebrate species probably has its origin in bird-assisted dispersal along the north-south mass migration routes.

5.6 Mammals

As everywhere in Europe, a megafaunal collapse has taken place during the late Pleistocene and early Holocene [113]. This remarkable species turnover is outstanding in the Aegean islands, where several endemics had developed, such as dwarf elephants [114]. Crete, for example, has one of the best-studied fossil histories of mammal colonization and adaptive radiation in the Mediterranean [115]. There is now much evidence that human-induced overkill is the reason for the rapid extinction of many animals on the Greek islands [116]. During the last 2000 years, several species have gone extinct on the mainland too; these include lion, bison, aurochs, and beaver, among others [117, 118] while the Anatolian leopard may have existed on Samos until little more than a century ago [119].

Greece's extant mammal fauna is still quite rich; it includes more than 115 species [80], although large mammals are noticeably scarce. The mammal fauna of exclusively semiaquatic or semiterrestrial species is limited, but many species utilize rivers, lakes, and wetlands. The European beaver (*Castor fiber*), a keystone "ecosystem engineer" in running waters, was native in mainland Greece and is said to have become extinct in the early nineteenth century, perhaps last recorded in the Alpheios river [117, 120]. However, this is difficult to verify, and there are conflicting interpretations of past ranges and exact extinction dates for the beaver in the Balkan countries [121]. Sadly, little has been written about the beaver in the Balkans or Anatolia, and a thorough review is much needed.

Mammals disperse rather slowly and are prone to extirpation and extinction, and they are limited by recently created natural and artificial barriers to movement. Extinction rates are more rapid on islands and peninsulas, and species may go extinct with little knowledge of how and exactly when (see references to large mammals on the Aegean islands in [119]). A semiaquatic mammal once thought to be rare and geographically localized in Greece is the Eurasian otter (*Lutra lutra*). In fact, it is quite widespread in mainland Greece but is rare and perhaps declining in xerothermic areas with little permanent water, although still found even on some islands, including very small river basins on Euboea, Kerkira, Lefkada, Samos, Chios, and Lesbos islands. The species may have existed on other islands, and there is anecdotal evidence for extirpation, but this is poorly documented. Interestingly, an endemic otter species (*Lutrogale cretensis*) existed until about the late Pleistocene on Crete; the genus *Lutrogale* survives in southern Asia but had a wider distribution in the past [114]. Terrestrial mammals do show some general biogeographic trends, some patterns being similar to the reptiles and amphibians, i.e., distinctive eastern elements in northeastern Greece (e.g., marbled polecat *Vormela peregusna* and Eurasian ground squirrel *Spermophilus citellus*) and the Eastern Aegean islands (e.g., some Asian bats, a squirrel, and certain rodents). In contrast, the mountains that run southward toward Greece assist in hosting many "temperate forest species" that are restricted to the cool forest habitats and not found in the central lowlands or the south of Greece. As birds, migrating mammals probably played a role as aquatic biota dispersers for short distances, especially in the distant past.

6 Charting Biogeographic Ecoregional Boundaries Using Indicator Species

The object of our biogeographic review here is to describe regional-scale patterns of aquatic/semiaquatic species distributions along with conditions that help chart discernible "freshwater ecoregional units." We do this from the perspective of the territory of Greece but necessarily must look further beyond the political boundaries to appreciate the "ecosystem region" patterns over wider areas. This work is largely based on an aquatic fish faunal survey of river basins [31], and as in the FEOW [9], the freshwater fish taxa are used as primarily biogeographic indicators.

6.1 Complexity and Uncertainty with Biogeographic Indicators: Freshwater Fish in a Mixed-Method Approach

Since it is well known that freshwater fish distributions often largely reflect historical patterns of river basin drainage connections, fish-based biogeographic investigations continue to develop in Greece and the surrounding countries. Important distributional and phylogeographic reviews have already been completed for most freshwater fish species in Greece (e.g., [58, 98]); some detailed molecular genetic approaches are also investigating fishes at the populations' level. No other wholly aquatic group of organisms is so well studied in the wider study area. Despite progress, there is still a need for taxonomic investigations and detailed distributional studies since many taxonomic revisions are taking place. Genetic-level screening of populations is also important in exploring relations among species that may have been translocated by humans. Fish must be used with care in biogeographic investigations.

The issue of scale is paramount in developing fish-based biogeographic interpretations. At the ecoregion scale, we are initially interested in interpreting the broad patterns, a continental-scale biogeography [2]. It is therefore permissible to initially compile information from so-called parent river basins (i.e., the largest hydrographic basin units), ignoring the many small basins that have the imprints of local or subregion ecological and idiosyncratic historic effects on the fauna (e.g., extinctions from stochastic events such as epic droughts, human-induced extirpations, etc.). Figure 8 outlines the biotically based inventory procedure for compiling each major river basin's fish assemblage (presence/absence data) for delineating freshwater biogeographic regions; Greece's 23 major "parent" river basins are mapped. Figure 9 shows typical cluster analysis classification of the species to assess the "parent" river basin similarities. The major faunal breaks are defined using the "parent" river basins [7, 36]. Figure 10 produces a different cluster dendrogram when 105 river basins are used in the classification analysis [37]. Despite differences in species sets used, the major biogeographic boundaries are retained. However, the latter cluster analysis does create unexpected patterns due to confounding effects such as a small number of variant basins. For example, the Thrace ecoregion is awkwardly clumped near the Southeastern Adriatic since only two basins from the latter are included (Prespa and Aoos) (Fig. 10). Arbitrary cutoffs in such classification analyses should by no means be used as a sole guide to chart biogeographic regionalizations. Finer-scale approaches and the use of other evidence to discern biotic distinctiveness should be applied.

Mixed-method approaches (integrating quantitative and qualitative information) based on various assumption-free numerical analyses and through building a broader evidence-based biogeographic understanding must help guide final boundary decisions. Investigating site-based samples of species abundance, instead of presence/absence data, may also be very informative [39]. Whatever the numerical analyses used in exploring interbasin faunal relationships, uncertainties and data gaps will be better interpreted through a mixed-method approach. Both quantitative analyses and expert judgment have guided the original FEOW [9] delineations and earlier works at the ecoregional scale [2].

7 The Current Regionalization Framework: A Review of the Freshwater Ecoregions of Greece

Here we provide an updated freshwater ecoregion map of Greece (Fig. 11) and a brief review of each ecoregion. This work is based on the FEOW [9] baseline and reviews of freshwater fish and aquatic and semiaquatic species distributions. This is

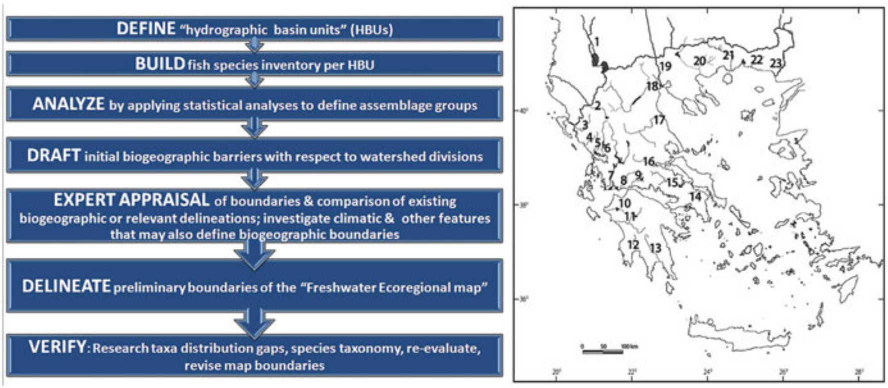


Fig. 8 Mixed-method procedure used to procure data, analyses, and delineations (*left*) and "parent" river basins which are larger than 700 km^2 and were used in initial analyses of fish-based classification and ordination: 1. Prespa, 2. Aoos, 3. Kalamas, 4. Acheron, 5. Louros, 6. Arachthos, 7. Acheloos, 8. Evinos, 9. Mornos, 10. Pinios (Peloponnese), 11. Alpheios, 12. Pamissos, 13. Evrotas, 14. Assopos, 15. Kifissos (Boeotian Kifissos-Kopais basin), 16. Spercheios, 17. Pinios (Thessalian Pinios), 18. Aliakmon, 19. Axios, 20. Strymon, 21. Nestos, 22. Filiouris, 23. Evros (adapted from [7])

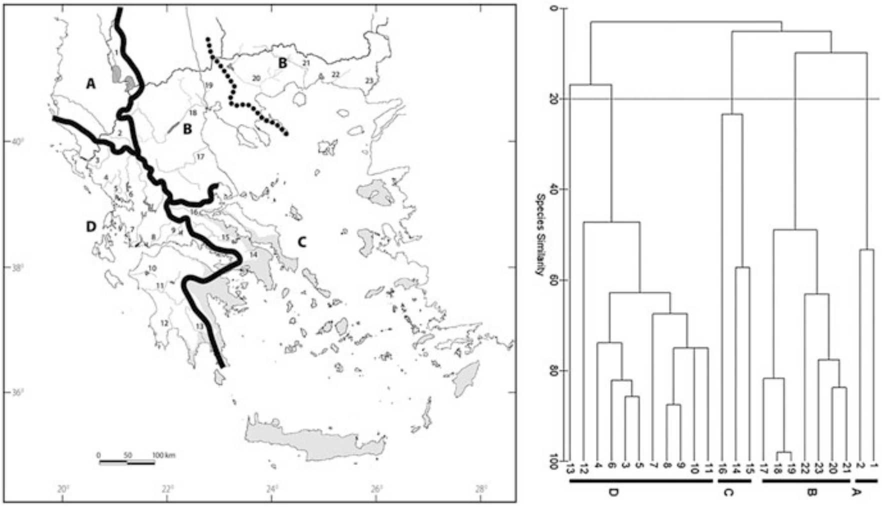


Fig. 9 An example of classification analysis of the major river basins based solely on fish species presence/absence (a subset of 90 lotic species). Cluster analysis (hierarchical clustering using group-average clustering from Bray–Curtis similarities) revealed four main biotically similar groups of basins at an arbitrary resemblance of 20%. This defines important faunal breaks on the watershed boundaries (*bold black lines*) of Greece's mainland. In this analysis lesser distinction is shown in the supra-region of the "Northern Aegean," area B split only by a *dotted line* (adapted from [7])

supplemented by knowledge of geological and climatological characteristics. We outline justifications for redefining boundaries and any changes made with respect to the FEOW [9]. Comparison among five important regionalization schemes is shown in Table 1, which tabulates the disparate biogeographic unit nomenclature as well. Efforts must be made to standardize the ecoregional/biogeographic regional names. We prefer to use the well-known anglicized names, instead of the local language-specific names in this review (i.e., Macedonia not Makedonia), but this has not held sway in earlier publications [14, 58].

7.1 Thrace

This is a species-rich region for aquatic biota; the so-called Thracian land bridge is located at the heart of the pivotal crossroads between the Balkans, Asia Minor, and the Black Sea Region. Freshwater biota has enriched the region due to a former connection with Black Sea and Danubian faunas (e.g., see [51]). It is an area rich in fish species, with at least 57 native species in the freshwaters of its Greek section [58]; large parts of this ecoregion expand into Turkey and Bulgaria. The Greek part hosts intriguing and specialized taxa such as lake-isolated shad species (*Alosa vistonica*, *Alosa macedonica*), migratory shemayas (*Alburnus volviticus*, *Alburnus vistonicus*), range-restricted loaches (*Cobitis puncticulata*), and Black Sea river gobies

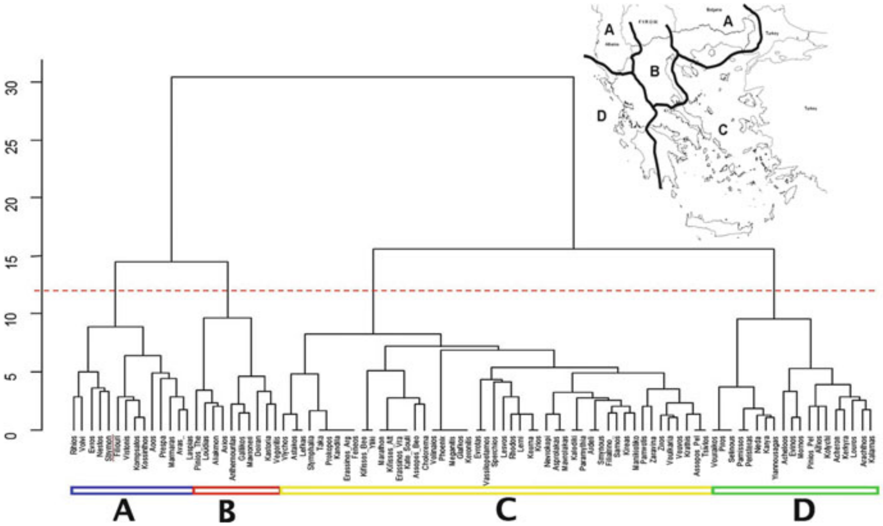


Fig. 10 Cluster analysis (Ward's method) with the presence/absence of all primary and primary-like freshwater fishes (120 native species) in 105 hydrographic basins. The arbitrarily defined cut-off at 12% creates four groupings: (**a**) "Thrace and Southeastern Adriatic", (**b**) Macedonia-Thessaly, (**c**) "Western Aegean and other islands," and (**d**) Ionian. The breakpoints between these groups are approximately projected on the *inset map* (adapted from [37])

(*Proterorhinus semilunaris*, *Neogobius fluviatilis*). Many of these species are related to species that are from the Danubian and Black Sea Region or are distributed further east in Asia Minor. Many fish genera are shared with the Macedonia-Thessaly region which also hosts Danubian species. Many other species of freshwater biota, including a freshwater shrimp (*Atyaephyra strymonensis*) and even a riparian willow (*Salix xanthicola*), are known to be restricted to the European part of this ecoregion. This richness is mirrored in terrestrial and semiterrestrial animals and many plants as well, which include terrestrial species from the Anatolian terrestrial fauna and flora as well, including some rare species restricted to this area within Greece [28, 122].

In Greece, the western limit of the ecoregion is a line of geologically old and stable mountains forming the watershed roughly between the Strymon and Axios river basin systems. The western boundary is enhanced by two idiosyncratic realms on this region's barrier line: the ancient lakes of the Mygdonia basin (Lake Koronia and Lake Volvi) and the rain shadow area and varied coasts of the Chalkidiki peninsula. The Mygdonia basin is a long-lived lake basin which functions as a refuge for various species from the Thracian ecoregion, while the Chalkidiki is a species-depauperate peninsula surrounded by deepwater areas creating a definite barrier to east-west freshwater species dispersal. The Chalkidiki peninsula is also a definite barrier for species dispersal between the Thrace and Macedonia-Thessaly Ecoregions

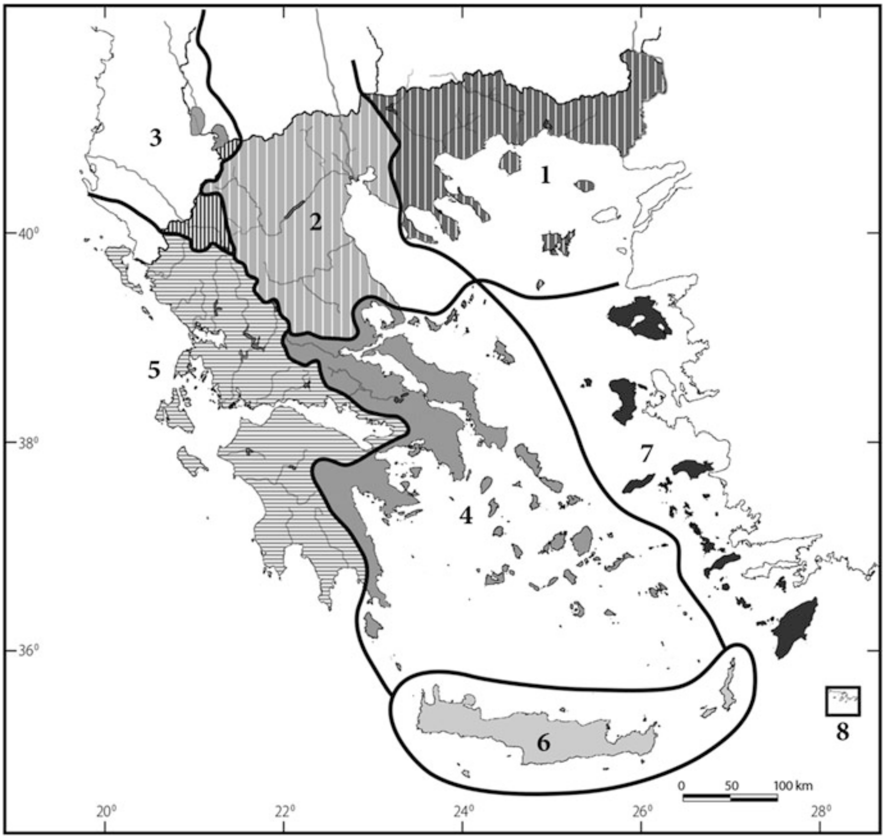


Fig. 11 Ecoregions of Greece, boundaries follow Zogaris [7] and the present study. Ecoregions *numbered* as in Table 1. See Fig. 2 for differences from the FEOW [9] of the study area

due to the deepwater marine areas around the peninsula (i.e., no significant confluences among major rivers where possible here during the Pleistocene sea-level transgressions) [123]. Despite this definite boundary area, in some fish-based biogeographic work, both Thrace and Macedonia-Thessaly have been charted together [124]; this supra-ecoregion has been called the "Northern Aegean."

There are still some poorly defined boundaries encompassing this region, especially outside the Greek territory. Bănărescu [14] charts the Thrace ecoregion exclusively in Europe, while Abell et al. [8] delineate it straddling both European and Asiatic shores of the Marmara Sea (Fig. 2). Since, during the last glacials, Thrace was fully connected with the northwestern basins of Asia Minor, the boundary of Abell et al. [8] should be justified; however, evidence for the specific boundary outline in Asiatic Turkey is required [125]. Finally the inclusion of the Northern Aegean islands of Limnos and Aghios Efstratios in this ecoregion should be better explored since they lie south of the North Aegean Trough which created a definitive deepwater

Table 1 The present study redefines some boundaries proposed initially FEOW [9] and reviewed in Zogaris [7]

	Present study	FEOW [9]	Bănărescu [14]	Maurakis et al. [15] (with subdivisions)	WFD 2000/60 ecoregions [4]
1	Thrace	Thrace	Thraki	Ponto-Aegean: Thracian-East Macedonian	Eastern Balkans (7)
2	Macedonia-Thessaly	Vardar	Macedonia-Thessaly	Ponto-Aegean: Macedonia-Thessaly	Eastern Balkans and Western Hellenic Balkans (7 and 6)
3	Southeastern Adriatic	Southeastern Adriatic drainages	South Adriatic-Ionian	Paleo-Hellas: Adriatic	Western Hellenic Balkans (6)
4	Western Aegean	Aegean drainages	Attika-Beotia	Paleo-Hellas: Attika-Beotia	Western Hellenic Balkans (6)
5	Ionian	Ionian drainages	South Adriatic-Ionian	Paleo-Hellas: Ionian	Western Hellenic Balkans (6)
6	Crete	Undefined region	Undefined region	Undefined region	Western Hellenic Balkans (6)
7	Eastern Aegean	Western Anatolia	Undefined region	Undefined region	Eastern Balkans (7)
8	Southern Anatolia	Southern Anatolia	Undefined region	Undefined region	Eastern Balkans (7)

The table compares nomenclature with four other ecoregion maps covering the study area. Some schemes had undefined areas, and these undesignated entities are mentioned. See Fig. 11 for delineations of the present study

zone separating the "European" continental shelf islands with the latter islands, which are geographically closer to the continental shelf area of the Troad of Asia Minor. Moreover, immediately to the south of the Troad, lies Lesbos, in the Eastern Aegean Ecoregion. Evidence-based guidance is needed to validate the proper ecoregional boundary around these islands since although the current boundary agrees with the phytogeography [17], others place a biogeographic boundary on the mid-Aegean Trough, i.e., between Samothraki and Limnos [22].

7.2 Macedonia-Thessaly

This diverse region is biologically closest to Thrace but with several important distinctive aspects and some different Danubian elements as well [33]. Its river basins include areas rich in wetlands and varied lentic surface waters, including several important ancient lakes, such as the Vegoritis basin lakes, Lake Kastoria and Lake Doirani. The last lake hosts at least one valid endemic fish species, and the former lakes have genetically distinct populations of fishes and invertebrate taxa. Like Thrace, Macedonia-Thessaly is also a fish species-rich area, hosting at least 49 native species in its freshwaters [58]. This rather small ecoregion hosts several range-restricted endemics, more than the Greek part of the Thrace ecoregion. This is due to the existence of more and older ancient lakes and to the unique geological history of the Thessalian Pinios basin as well [14]. The Pinios lowlands were a former lake bed, and it hosts three local endemic fish species (*Cobitis stephanidisi*, *Knipowitschia thessalus*, *Gobio feraeensis*) and several other genetically distinct fish taxa (i.e., local variant of *Barbus macedonicus*, etc. [58]). Paleogeographic research in the Thermaicos Gulf has shown that despite these distinctions, the Pinios was a tributary of the greater Axios paleoriver about 24,000 years ago [126].

As explained in Sect. 7.1 "Thrace," Thrace and Macedonia-Thessaly have been shown to be ichthyogeographically closely related in numerical taxonomic classifications [124]. When solely considering riverine fishes, a relatively modest ichthyogeographic difference is detected between the Axios (Macedonia-Thessaly) and Strymon (Thrace) [36](Fig. 9). However, this close relation requires further investigation. If one adheres to the FEOW delineation of Thrace, which includes territory in NW Asia Minor, the ichthyological distinctions will carry more weight (i.e., if more river basins are included in the analysis). Nearly all holistic biogeographic publications support the ecoregional boundary between Macedonia-Thessaly and Thrace [7, 8, 14, 37, 39]. A frequently stated biogeographic hypothesis for such a different assemblage in Macedonia-Thessaly is that the Axios valley functioned as a dispersal "roadway" leading south for Danubian Morava-origin rheophilic fish and invertebrate species [14], and this is a key distinction between this ecoregion and Thrace for its fish fauna [31]. The relationship and possible biotic connections between the Axios and Danubian Morava may be explained by possible episodes of "river capture" (river piracy) since the headwaters of these rivers are in close proximity within Southern Serbia. Recent reviews show that new fish species splits are eminent in the Macedonia-Thessaly region, and new interpretations of the ichthyofaunal provenance of some species should prove the area as being even more ichthyologically distinct in the near future (see [58, 98]). This complex fish-based boundary controversy shows the importance of providing mixed-method evidence for delineations in precautionary incremental steps.

Finally the name Macedonia-Thessaly was proposed by Maurakis et al. [15] and Bănărescu [14], and we use this instead of "Vardar" as proposed in FEOW [9]. Vardar is a Slavic name for a single river (Axios/Vardar) in this otherwise very diverse and culturally prominent region; we feel it is proper to use the classical geographical names that best help distinguish the region.

7.3 Southeastern Adriatic

Although most of the region lies within Albania, parts of the Aoos and Prespa Lake tributary basins are in Greek territory. This ecoregion is one of the most endemic species-rich; however, its aquatic species taxonomy, phylogeography, and general biogeography are poorly studied. This ecoregion holds a long-isolated fish species assemblage and one that has evolved and diverged due to important vicariance events and the existence of long-lived lakes (Ohrid, Prespa, and Skadar) [14]. The region is rich in fish species [127]; about 20 native freshwater species are recorded in the part of these basins that belong to Greece [58]. There are only very few affinities with the Thrace and Macedonia-Thessaly regions (i.e., some Danubian genera that have since diverged [98, 103]). This region has remarkable differences with the adjacent Ionian region to the south; however it has many "shared species absences" with the Ionian since relatively few Danubian fish species exist in both the Southeastern Adriatic and Ionian. The region also has a higher percentage of regionally endemic fishes than Macedonia-Thessaly or Thrace.

The southern boundaries of this region are fairly distinctive. A rather abrupt faunal break exists in the southern rim of the Aoos basin watershed divide, i.e., between the Aoos and Butrint/Kalamas basins. The Butrint basin includes the Bistrica river, located just north of the Greek-Albanian border but belonging to the Ionian ecoregion. This boundary was originally charted by Bianco [128], but it was not identified in a later descriptive analysis [33]. Bănărescu [14] showed this boundary on a map, but in his detailed description, he defined a broader regional unit, the "Ionian-South Adriatic," as a single biogeographic region. Zogaris et al. [36, 37], Oikonomou et al. [124], and Economou et al. [39] confirm the Butrint/Kalamas-Aoos boundary (although it was erroneously charted in the FEOW (2009) map (Fig. 2)).

The coastal marine depths between the Aoos and Butrint basin help define the southern boundary. The steep sloping continental shelf along the southern Albanian coast (Strait of Otranto) effectively separates the Butrint from the Aoos basin, presumably even during sea-level regressions which took place during the Tertiary and Quaternary [123]. It is possible that the Southeastern Adriatic biogeographic unit has been influenced by incursions of species from the Danube due to river piracy among the Danubian Sava and Drin tributaries. This makes it distinctly different from the Northern Adriatic also [124].

Finally the ancient lakes of Ohrid and Prespa represent a world-renowned endemicity hotpot, and they are biogeographically related to the faunas of the north and south of Albania, respectively [127]. Phylogeographic relationships of several freshwater fishes of Prespa with the southern Albanian drainages (formerly endemics of Prespa) have been recently clarified [129]. The unresolved taxonomic status of several fishes in the area and poor knowledge of their distribution have created difficulty in charting biogeographic boundaries. We therefore propose one should not consider the Prespa lake basin (or a wider entity including Ohrid) as a distinct biogeographic region as has been published by Oikonomou et al. [124]. Ohrid has a higher overall endemicity than its shallower sister Lake Prespa, and their faunas are remarkably different [130]. Both lakes as is Lake Malik are hydrologically connected to the Adriatic. Finally the Prespa basin area, which has an areal cover of about 2,520 km^2, does not constitute a large enough areal entity to be considered an "ecoregion" (as compared to other biogeographic regional units or the freshwater ecoregion areas [9]). Nevertheless, Lake Prespa and Lake Ohrid are globally important biodiversity hotspots and should be considered distinct parts of the Southeastern Adriatic freshwater ecoregion.

7.4 Western Aegean

This is a rather small freshwater ecoregion and the most geographically fragmented such entity in Greece. For a long time, the eastern coast of mainland central Greece south of and including the Spercheios basin was called the Attiko-Beotian region [15, 131]. The FEOW [9] expands this region; it now includes the Western Aegean islands (Cyclades, Euboea, Northern Sporades) and the northeastern parts of the Peloponnese peninsula. Zogaris [7] corroborated the general regionalization but modified these boundaries to follow watershed lines belonging exclusively to the Aegean basin, as presented in Fig. 11. This ecoregion includes an extensive rain shadow area that sustains a seasonally semiarid area with frequent prolonged droughts (this is a climatically homogenizing effect created by the Pindus cordillera). Most of the area is made up of dry-land calcareous mountainous landscapes with rather scarce running water ecosystems and many small-sized seasonally arid river basins. Although the area's "small waters" (i.e., springs, rivulets, etc.) host many interesting local endemics – especially of smaller aquatic animals such as spring-inhabiting aquatic snails [75–77]– the region is generally considered species-poor for aquatic life. Although this is so for fishes and many larger aquatic plants due to the scarcity of larger permanent waters, its aquatic invertebrates are poorly studied and may include many undescribed taxa [76, 91].

This is a difficult ecoregion to delineate. Some boundaries are still poorly justified in a biological sense [7]. The region hosts about 30 native fish species in its waters and most are actually widespread marine migrants and transients; some species need further taxonomic research, including the *Alburnoides* and *Rutilus* of the Spercheios, *Squalius* of Euboea, *Aphanius*, and *Knipowitschia* [58]. This geographically fragmented complex of peninsulas and islands hosts only three major "parent" river basins holding substantial fish faunas (i.e., Spercheios, Kifissos-Kopais, and Assopos). The region sustains very few lakes, one of them a marshy extensive ancient lake area, the greater Kopais basin (including the Kifissos and Lake Yliki and Lake Paralimni) in Boeotia. This is the region's most endemic-rich area with emblematic fishes such as *Telestes beoticus*, *Scardinius graecus*, and *Rutilus ylikiensis*, while the enigmatic *Luciobarbus graecus* is shared with the Spercheios. Further north, the river Spercheios is exceptional since it is a biogeographic crossroads also hosting several fish genera from the Macedonia-Thessaly freshwater ecoregion [39]. Fish phylogeography also provides evidence for a connection between the Spercheios and the Pagasitikos Gulf to the north (the Pelion peninsula being a biogeographic barrier).

Ecoregional boundary justification becomes difficult especially in the southern part of this region. The remarkably dry limestone landscapes of the eastern Peloponnese, the Saronic and Argolic Gulfs and the Cycladic islands, have very few stretches of perennial streams or other permanent water; this is one of the driest parts of the country. The eastern Peloponnese is also a rain shadow area with many "shared species absences" with the Attiko-Beotian heartland. In this way the eastern Peloponnese is similar to the other Western Aegean drainages, but the boundary lines are difficult to set precisely. Taxa-particular distributional idiosyncrasies complicate boundary issues, since in some parts of this region, as in the Ionian ecoregion, many spring-inhabiting endemics thrive [95]. Many zoologists have considered the Peloponnese a distinct biogeographic unit (see [7]), although some who study aquatic biota have also provided evidence for an east-west split (e.g., [26]). The east-west boundary within the Peloponnese, as prescribed in this study, requires further research.

This region includes the Cyclades islands which during the Pleistocene glaciations were much larger "mother-island" entities surrounded by smaller islands. The Cyclades were also probably connected via Attika and Euboea, during the middle Pleistocene, ca. 180–140 kya BP [132]. Despite their current aridity, the Cyclades surprisingly host several endemic aquatic insects [91, 133] and many terrestrial invertebrates as well [134]. Euboea is Greece's second largest island, a true continental island that was connected to Boeotia and Attica during the early Holocene, a few thousand years ago. However, Euboea's long and rather high mountains are geographically and geologically isolated, creating river basins that are long independent from the mainland's basins. This long mountainous island is unusual in being relatively rich in significant spring-fed perennial flowing streams that also host endemic macroinvertebrates [75] and even two endemic fish taxa, an undescribed chub *Squalius* sp. and the critically endangered Evia barbel *Barbus euboicus* [58].

Since the Western Aegean's aquatic animals are still rather poorly inventoried [14, 84, 91] and perennial surface water features are patchy and isolated, the particular biogeographic boundaries have never really been precisely charted. The delineation of the "Aegean drainages" freshwater ecoregion by FEOW [9] is unfortunately inaccurately and coarsely charted since it erroneously includes basin areas that have no biogeographic relationship to this particular region (i.e., the northern Corinthian Gulf drainages which obviously drain into the Ionian basin and parts of Thessalian Northern-Pelion and Mavrovouni mountains which have relations with the fauna of northern Greece) (see Fig. 2). Therefore, this ecoregion's final delineation is obviously in need of documented evidence in order to verify the redefined boundaries in our revised map (Fig. 11).

7.5 Ionian

Greece's west coast surface waters are collectively the most endemic species-rich freshwater ecosystems in the country. A small part of this ecoregion also belongs to the southernmost part of Albania (the Butrint basin and Bistrica river, south of the Aoos watershed). Many endemic fish and other aquatic animals are extremely range restricted in this, or in isolated parts of this, ecoregion [31]. The region has about 48 native fish species in its freshwaters [58]; however, basin-scale species richness is much poorer than Macedonia-Thessaly or Thrace [39]. In mainland Greece native fish distributional patterns clearly indicate that the Pindus mountains create a prominent long-term biogeographic discontinuity that separates distinct freshwater biogeographic regions east and west of the Pindus, something well depicted in many biogeographic studies of the wider region [33, 124]. The Pindus biogeographic barrier has been corroborated by the distributions of many animals and plants including selected amphibians [107]; semiaquatic terrapin; *Emys orbicularis*, among other species [135]; floristic assemblages [60]; and selected aquatic invertebrates, such as the Hydrobiidea gastropods [14] and freshwater shrimp [85]. Despite its remarkable isolation by the ribbonlike Pindus mountain chain, there is much variation within this ecoregion.

One of the most outstanding aspects that characterize this ecoregion is the remarkable diversity at the subregional level. Parts of the region have unique water bodies, isolated in the past by former inland lakes, such as the Corinthian Gulf, or long-standing fairly large river basins with extant long-lived lakes, such as the Acheloos. The lotic waters around the ancient lake of Trichonis are especially interesting [136], and new species are still being described in this global biodiversity hotspot. Distinctive and long-isolated basins such as the Evrotas, for example, are unique and idiosyncratic for their range-restricted endemics and depauperate biocommunities, well adapted to non-perennial and spring-fed stream conditions [57]. Of the river basins included in this heterogeneous ecoregion, the so-called northern Ionian (northwest of the Acheloos watershed line) seems to create a distinctive subregional entity. Many species of fish are restricted to this area, including distinct species of the genera of *Pelasgus*, *Squalius*, *Telestes*, *Knipowitschia*, *Valencia*, and *Cobitis*. Work on aquatic invertebrates in the Ionian ecoregion will reveal very interesting biogeographic interpretations, including new species and informative subregional patterns; a productive area is research on freshwater mussels [137], freshwater shrimps [85, 138], and fish parasites [59]. More research into the fishes and other aquatic life of the Ionian ecoregion is required, and new species will certainly be described in this relatively understudied "center of endemism."

All the offshore Ionian Islands are definitely a part of the Ionian ecoregion, and they constitute a species-depauperate area compared to the varied biotic riches of the adjacent mainland. An exception to this is Kerkira, which is really a recently isolated continental island, being connected to the mainland 8,000 years ago [50]. The island waters were obviously connected to the Kalamas and Butrint basin, and today the island still sustains diverse stream and wetland ecosystems [104, 139]. As would be expected, the Ionian Islands' aquatic fauna is very similar to the mainland. For example, the Ionian Islands share a similar caddisfly fauna (Trichoptera) with the mainland's west coast, in contrast to the many endemics found in the Aegean islands [81]. Similarly, the flora of the islands also has more connections to the mainland than in the endemic-rich Aegean [140]. Even some species of isolated fish are genetically closely related to the west coast mainland species [141], and there are no endemic fish species known to be restricted to any of the islands [104].

7.6 Crete

Crete has been called a small "continent" due to its distinctiveness and diversity [142]. Van der Geer et al. [114] refers to Crete as an "oceanic-like island" for its unique geological and zoogeographical history. This proposed freshwater ecoregional unit includes Crete and its satellite islets and the Karpathos archipelago (with three main high-relief islands and about 20 very small islets). Controversy about the biogeographic ecoregional status of Crete and its surrounding islands persists.

Phytogeographically, Crete is a separate floristic region, noted as "Kriti + Karpathos" in the Flora Hellenica map [17, 60]. With respect to the Karpathos archipelago cluster, Raus [143] concluded that botanic relations are to the west, to the "European" Cretan and south Aegean flora, rather than to the Asian flora of the other Dodecanese Islands. The Karpathos archipelago has several faunal elements not found in Crete but exhibits many "shared species absences" relative to the species-rich Dodecanese, which are nearer to the Asian coast. However, nearly all zoologists traditionally group the Karpathos islands toward the Asian Dodecanese and not with Crete. The Karpathos archipelago has been geologically isolated from the Cretan landmass for over 10 MYA; however, inclusion within a broader Crete ecoregion is justified based on the internal endemism seen among the different former "paleo-island" sectors of Crete. In this way the Crete Freshwater Ecoregion can be conceptualized as an "insular" ecoregion linking isolated but geographically proximate and geologically similar areas together. The FEOW [9] global freshwater ecoregion delineation did not classify the Cretan area to a particular ecoregion; Crete along with Karpathos-Saria-Kassos was labeled as "undesignated" on the global FEOW map. During the development of the expert-guided process of the FEOW project, there were differing opinions about the place given to the island, and one of the early maps had erroneously lumped the island to Southern Anatolia and Cyprus (Abell, R. pers. com). In contrast, Crete is a distinct ecoregion in the terrestrial ecoregions of the world map but without the Karpathos archipelago [11, 45] (Fig. 5). Crete was proposed as an independent freshwater ecoregion in Zogaris [7], and in this account we suggest a distinct freshwater ecoregional status for Crete and the Karpathos archipelago.

The wider geological context here is important for interpreting the proposed ecoregion's boundaries and its relationship and affiliation to other geographic areas [114, 144–146]. For nearly six million years, Crete have been completely isolated from the continent. About 15 MYA Crete belonged to a large subcontinent that extended from the Western Balkans to Asia Minor. The Aegean landmass subsided under the sea beginning 10 MYA, and only the higher mountains, the so-called Cretan paleo-islands, remained above water. This period of vicariance within the island chain augmented evolutionary development in many species [62, 83]. The Karpathos archipelago was also isolated from Asia, long before Rhodes, and it has been isolated from Rhodes for at least 3.5 MYA [145]. Around 2 MYA, the wider region of Crete was tectonically uplifted and much land emerged, and the palao-islands were joined to form Crete's present-day outline. Crete is characterized by an impressive and ancient mountain range system with four steep massifs and more than 20 satellite mountains. Karpathos, Saria, and Kassos although very narrow and steep islands also have very high mountain ridgelines and gorges similar to Crete.

Climatic conditions and hydrology also connect Crete and the Karpathos archipelago. The eastern part of Crete and the Karpathos archipelago are located within a prominent rainshadow (created by the Cretan mountains), and this produces some of the driest conditions in Europe in Eastern Crete [142]. Crete's running waters are diverse, but many are non-perennial flowing and subterranean; surface water ecosystems are stressed due to both high evapotranspiration rates, climatic variability, and widespread human-induced water over-abstraction [147]. Naturally arid ephemeral streams dominate in Eastern Crete and the Karpathos islands, and some are nearly like semidesert wadis. The eastern part of Crete and the Karpathos area are a very windy part of the Aegean compared to the more leeward relatively calm conditions on Rhodes [148]. In this way climatically and geologically, the Karpathos archipelago shows more affinity with Eastern Crete than the more humid conditions and gentle landscapes of Rhodes.

Since geologically isolated islands such as Crete are prone to natural and anthropogenic extinctions, these islands have very few native fish species. Crete is definitely one of the poorest ecoregions for native fish fauna in the Mediterranean as nearly all native fishes in inland streams and lakes are of marine origin (about 11 native species). Actually only a very few native species live all their life cycle in inland waters, namely, the localized river blenny *Salaria fluviatilis* and landlocked smelt *Atherina boyeri* [149]. For this reason, invertebrates, the fossil record, and perhaps water plants should be better explored for the region's freshwater biogeographic description. Although, endemism among the extant terrestrial plants and invertebrates is impressively high, most aquatic species distributions are not well studied [42]. Also, the aquatic biodiversity of Crete is probably characterized by the extirpation of several aquatic species due to modern anthropogenic wetland and surface water degradation [65, 142].

Today, Crete and the Karpathos archipelago certainly have a depauperate freshwater biota relative to the large continental islands or peninsulas of the Balkans and Asia Minor, but there is a relatively high number of endemic freshwater and wetland invertebrate life forms, including endemic aquatic insects (e.g., Trichoptera, Coleoptera, Heteroptera, Plecoptera, among other groups), freshwater crabs, a freshwater shrimp, and endemic amphibians. Because the Crete ecoregion is a long-isolated "former" island chain, its overall uniqueness is well known [146, 150], but further study of its aquatic and wetland species is definitely required for a complete biogeographic review. Finally a thorough evaluation of the freshwater biogeographic affinity among Crete and the Karpathos archipelago must be researched. For now, we take a precautionary approach and suggest union of the Karpathos archipelago with Crete.

7.7 Eastern Aegean

Most of the Greek islands in the Eastern Aegean show biological affinities to the faunal assemblages of Asia Minor, but freshwater assemblages are comparatively species depauperate to the adjacent Asian continent. Unfortunately, the aquatic biota remain poorly studied, and new species have been recently recorded and described in the area, such as on the wetland-rich Lesbos island [151]. The Eastern

Aegean islands have few freshwater fish; although at least 18 native species inhabit freshwaters, only six are confirmed primarily freshwater fishes. All native primary freshwater species show strong affinity to Asia Minor [58, 152]. Abell et al. [8] correctly regard the Eastern Aegean islands as part of their so-called Western Anatolia Freshwater Ecoregion (see Table 1); other researchers use the name “Eastern Aegean” to be consistent with marine area geographic terms often used in other regionalizations [35].

The literature provides full biogeographic support to the notion that these insular ecosystems should constitute part of an ecoregion that belongs to western Asia (Anatolia) [119, 153–155]. Botanists, such as Strid [17], call the biogeographic line at the mid-Aegean trench “Rechinger’s line” which in his words “constitutes the phytogeographical borderline between Europe and Asia” in the Aegean. However, there are varied patterns in the aquatic biota distributions among the different islands. Some islands such as Tilos, Fournoi, Nissiros, and Kalymnos, for example, have very little surface water, and their freshwater biota is relatively unknown. Some islands were rather recently connected to the Asian mainland less than 10,000 years ago (Lesvos, Chios, Samos, Kos), while others, such as Rhodes, had separated approximately three million years ago [145].

Rhodes is especially interesting due to its long-term insular isolation, its southern location near the Southern Anatolian shores, and rather diverse stream ecosystems. Although the island has a unique endemic fish (*Ladigesocypris ghiggi*), it also sustains a species of freshwater shrimp (*Palaemon colossus*), which it shares with the Southern Anatolian Ecoregion [85]. Rhodes is also remarkably isolated from the southernmost of the Dodecanese Islands, Karpathos, and Kassos – totally separated perhaps for at least 3.5 MYA [24] – while these two islands are attributed by phytogeographers to the Crete ecoregion. Baseline species inventory work and research are urgently required on both Eastern Aegean islands and the adjacent Anatolian shores in order to explore freshwater biotic relations and provide for potential subregional delineations and a thorough biogeographic interpretation.

7.8 Southern Anatolia

Greece administers a tiny island cluster of just 11 rocky islets and sea rocks, the Kastellorizo cluster, along the Mediterranean coast of Southern Anatolia. This island cluster lies just 2 km off the Anatolian shore and lies east of the Western Aegean-South Anatolia division boundary as charted by Abell et al. [8] and other terrestrial and aquatic biogeographic delineations of Asia Minor (e.g., [125, 156]). The Kastellorizo island group has no large wetlands apart from tiny micro-springs and temporary pool-like depressions, sometimes flooded in the winter season; the stream courses are only ephemeral gullies, and there are a few artificial cisterns, wells, and a tiny modern reservoir [157]. The tiny artificial water bodies hold a very few hydrophilous plants [158]. Of course, no native freshwater fishes exist on the island. It does host a terrestrial amphibian, that is definitely a species of Southern Anatolia, the

Lycian salamander (*Lyciasalamandra luschani*) (Fig. 7), but we have no knowledge of its hydrophilous invertebrate fauna in its few aquatic/semiaquatic ecosystems. The biogeographic boundaries of Southern Anatolia are not well explored [125]. Research is needed to confirm the Eastern Aegean/Southern Anatolian ecoregional boundary and to better define the latter.

8 Discussion

8.1 Freshwater Ecoregions as a First-Tier Biogeographic Framework

The current freshwater ecoregional map of Greece provides a holistic regionalization framework, grouping river basins based on major biotic similarities and relevant geological and climatic attributes. As Bailey [1] has said: "such exercises in regionalization approach truth by a series of approximations." Despite this current map's mixed-method approach and expert-guided procedure with the associated caveats [7], the ecoregional units have already been very useful in inventory and conservation research (e.g., [39, 58, 159]). The freshwater ecoregion map is suited for in-depth, intraregional analysis that could better support boundary validation and the definition of biogeographic "subregions" in order to further assist river basin classification [160]. Finally, we reiterate Forman's [161] wise words: "Regional ecology is a little-understood research frontier that will noticeably strengthen conservation, planning, sustainability, and land-use policy. We had better learn the ecology of regions."

8.2 Taxonomic Complexities and the Taxonomic Impediment

Freshwater fish distributions have traditionally guided aquatic biogeography. One current problem with solely using fish in regionalizations is taxonomic. An unprecedented percentage of European fish species name changes has taken place in the last two decades [127]. Most of the changes have resulted from the application of new taxonomic concepts and methods, especially the adoption of the phylogenetic species concept (PSC), which has now replaced the biological species concept (BSC) (for a review, see [31]). Traditionally, under the BSC, a "species unit" is a group of actually or potentially interbreeding populations. The PSC, by contrast, considers "species" as the smallest diagnosable cluster of individuals within which there is a parental pattern of ancestry and descent. In this context, the PSC accepts the evolutionary potential of a lineage that has just started to separate from other lineages as the main criterion for defining species. Thus, under the PSC, there are no subspecies. As a result, many taxa recognized as subspecies under the BSC have often been raised to the species rank. As taxonomic research continues, former species and subspecies will either tend to be "split" into distinct species or "lumped" within already valid species [141]. Several new freshwater fish species are expected to soon arise through this research within the area of Greece's territory [58]. Ongoing research in the fish populations' systematics and phylogeography will continue to guide constituent freshwater biogeographic boundaries.

These taxonomic complexities obviously spread to all species, not just fish. If current species distributions are to be utilized in biogeographic analyses, adequate taxonomic and phylogeographic information must be inventoried, organized, interpreted, and published. Part of the reasons for very different approaches to freshwater biogeographic delineations is that Southeastern Europe's species-level taxonomy is still far from being resolved, and there are still data gaps in basic biodiversity distributional knowledge [41]. There is no better time to restate the value of organizing a broadscale effort for a full biodiversity inventory of inland waters. We believe foreign and Greek professional researchers and amateur naturalists should coordinate and participate in collections in Greece. The "taxonomic impediment," reflecting a global shortage of taxonomists and systematists which negatively impacts biodiversity conservation, is a worldwide problem [162]. Organized biological collection campaigns, taxonomy, and phylogeography should become defining priorities for conservation-relevant aquatic research in Greece and other Mediterranean countries [163].

Finally, a rising issue in biogeography is xenobiodiversity, the increase and spread in non-indigenous species that are artificially dispersed by humans. Here too, due to the taxonomic impediment, this issue has been poorly monitored in inland waters in Greece [164]. This concerns both alien species from abroad and locally translocated species from nearby ecoregions. In some cases, it is difficult to be sure if certain populations are translocated or have a naturally disjunct distribution. An example is the unusual Caucasian goby (*Knipowitschia caucasica*) population in a coastal stream near Karystos on Euboea Island; its genetically closest relatives are in the Thracian Ecoregion [165]. Since a genetic screening of other Western Aegean *Knipowitschia* gobies has never been done, we cannot be sure if the Karystos population is natural or introduced by humans. Evidence for species translocations across ecoregions, involving both vertebrates and invertebrates native to Greece – but not indigenous to the river basin areas they currently inhabit – is a serious conservation concern (e.g., [84, 159]). Monitoring for biodiversity must necessarily study xenobiodiversity to interpret patterns, trends, and impacts to local biodiversity.

8.3 Freshwater Ecoregion Delineation Difficulties

Hartshorne [166] described the concept of region as being characterized by "relative homogeneity in prescribed characteristics, selected for their salience in highlighting areal differences at the regional scale." In Greece, it has been repeatedly shown that freshwater fish assemblages can effectively depict dispersal barriers and the influence of hydrographic history and paleogeography [15, 34]. However, fish-based classification of river basin relatedness does have limitations in species-depauperate areas dominated by very small basins and islands [36]. These challenges are instrumental in terms of identifying basic needs for evolving better methods to delineate ecoregion boundaries [2, 44, 167].

Greece and the surrounding Eastern Mediterranean countries are also challenging areas for tracing biogeographic patterns at the regional scale because of the intensive influence humans have had on ecosystem modification and cultural landscape patterns [116]. In zoogeographic and vegetational sense, this has been most intensive in the southern half of Greece and the islands [116, 168]. Species-depauperate conditions in some areas of the south may point to recent anthropogenic extirpation instead of "natural" biogeographic patterns; a potential example of this is the species-poor riparian zones in more populated and degraded river basins of southern Greece [70]. Humans have also transported/translocated many species, especially on the islands (e.g., [116, 169]), in reservoirs, and in larger river basins [159]. Humans have shaped the landscape patterns in such a way as to sculpt the evolution of so-called cultural landscapes, where human-modified habitat types now dominate [168]. In the Greek islands, human influence has been widespread for at least 8,000 years [48, 142].

It should be made clear that on the islands and peninsulas, there are naturally increased extinction rates [108]. Detailed work to define exact ecoregional (and subecoregional) boundaries is needed in the islands and the southern half of Greece's mainland due to the natural (or human-induced) species-depauperate conditions. Macroinvertebrates are important target groups for biogeographic research here, and their communities may differ markedly from adjacent mainland conditions, e.g., the near-natural streams of Samothraki Island [170]. Many small streams in the islands may sustain macroinvertebrate communities that have survived totally isolated for millions of years [133], and many of these are still poorly explored [14, 42, 171]. Inventory and taxonomic work will provide scientific justification for the legislative conservation of the so-called small waters of the islands and xerothermic Mediterranean coasts; many of these areas' small streams and wetland conservation values have long been underappreciated [76, 163].

9 Conclusions

Ecoregional maps may inspire controversy, but they are also powerful organizational, educational, and exploratory tools. Charting biogeographically based freshwater ecoregional units has been especially challenging in Greece since tectonic, climatic, sea-level, and anthropogenic changes have created outstanding complexity. The eight freshwater ecoregion delineations that encompass Greece's territory are akin to the "Freshwater Ecoregions of the World" [9] delineations, but specific boundaries have been redefined in this review. We strongly suggest a revision of the FEOW [9] boundaries based on the incremental revisions in this study for Greece.

Although Greece is a living laboratory for biogeographic studies, many of the aquatic biota have been poorly studied. Extensive aquatic and semiaquatic species distributional inventories will be required for a full review and more complete interpretation of freshwater biogeographic patterns. Particular emphasis must be given to key indicator aquatic groups for inland waters, such as the EU WFD's biotic quality elements: fishes, benthic macroinvertebrates, and aquatic plants. This will help couple policy-relevant EU water management with biodiversity conservation initiatives. A detailed fish atlas (and associated archive of specimen and genetic collections) is an imperative for furthering any kind of organized fish-based biogeographic and phylogeographic work. Fish populations require genetic screening, and novel molecular analytical methods now help speed up the inventory process. Currently, data on the taxonomy and precise distributions of aquatic macroinvertebrates is particularly poorly developed in Greece, and these groups are highly important aquatic biogeographic indicators. Researchers from Greece and other countries must cooperate to increase the intensity of organized field collections. Biogeographical research should provide an impetus to coordinate more productive taxonomic and phylogeographic research that will ultimately assist scientifically guided conservation actions.

Acknowledgments We dedicate this work to Professor Anastasios Legakis (University of Athens) who has been a careful and meticulous "curator" of zoogeographical research in Greece for a very long time. We thank all colleagues at the Inland Waters section of the Institute of Marine Biological Resources and Inland Waters at HCMR who contributed in various ways to data archiving, data quality control, and biogeographic discussions. We especially thank Yorgos Chatzinikolaou for his help in the early part of this work, and we are grateful to Yorick Reyjol for the important assistance in statistical analyses.

References

1. Bailey RG (1998) Ecoregions: the ecosystem geography of the oceans and continents. Springer, New York, NY
2. Abell R, Thieme M, Dinerstein E, Olson D (2002) A sourcebook for conducting biological assessments and developing biodiversity visions for ecoregion conservation. Freshwater ecoregions, vol II. World Wildlife Fund, Washington DC
3. Turak E, Dudgeon D, Harrison IJ, Freyhof J, De Wever A, Revenga C, Garcia-Moreno J, Abell R, Culp JM, Lento J, Mora B (2017) Observations of inland water biodiversity: progress, needs and priorities. In: Walters M, Scholes RJ (eds) The GEO handbook on biodiversity observation networks. Springer, Cham, pp 165–186
4. European Commission (2000) Directive 2000/60/EC, Establishing a framework for community action in the field of water policy. Off J Eur Comm L 327:1–71. Brussels
5. Gordon ND, McMahon TA, Finlayson BL, Gipple CJ, Nathan RJ (2004) Stream hydrology: an introduction for ecologists 2nd edn. Wiley, Chichester
6. Montgomery DR, Buffington JM (1998) Channel processes, classification, and response. In: Naiman RJ, Bilby RE (eds) River ecology and management: lessons from the Pacific Coastal Ecoregion. Springer, New York, NY

7. Zogaris S (2009) Contribution to a biotic classification of the rivers of Greece based on ichthyofauna and riparian vegetation. Doctorate Degree in Environmental and Natural Resources Management, University of Ioannina, Agrinio. (in Greek)
8. Abell R, Thieme ML, Revenga C, Bryer M, Kottelat M, Bogutskaya N, Coad B, Mandrak N, Balderas SC, Bussing W, Stiassny ML (2008) Freshwater ecoregions of the world: a new map of biogeographic units for freshwater biodiversity conservation. Bioscience 58(5):403–414
9. FEOW (2008) Freshwater ecoregions of the world-conservation science program. The Nature Conservancy and World Wildlife Fund, Washington DC
10. Bailey RG (2005) Identifying ecoregion boundaries. Environ Manag 34:14–26
11. Olson DM, Dinerstein E, Wikramanayake ED, Burgess ND, Powell GVN, Underwood EC, D'Amico JA, Itoua I, Strand HE, Morrison JC, Loucks CJ, Allnutt TF, Ricketts TH, Kura Y, Lamoreux JF, Wettengel WW, Hedao P, Kassem KR (2001) Terrestrial ecoregions of the world: a new map of life on earth. Bioscience 51:933–938
12. Omernik JM (2004) Perspectives on the nature and definition of ecological regions. Environ Manag 34(1):S27–S38
13. Holt BG, Lessard JP, Borregaard MK, Fritz SA, Araújo MB, Dimitrov D, Fabre PH, Graham CH, Graves GR, Jønsson KA, Nogués-Bravo D (2013) An update of Wallace's zoogeographic regions of the world. Science 339(6115):74–78
14. Bănărescu PM (2004) Distribution pattern of the aquatic fauna of the Balkan Peninsula. In: Griffiths HI, Kryštufek B, Reed JM (eds) Balkan biodiversity: pattern and process in the European hotspot. Kluwer Academic Publishers, Dordrecht
15. Maurakis EG, Pritchard MK, Economidis PS (2000) Historical relationships of mainland river drainages in Greece. Bios 6:109–124
16. Lymberakis P, Poulakakis N (2010) Three continents claiming an archipelago: the evolution of Aegean's herpetofaunal diversity. Diversity 2:233–255
17. Strid A (1996) Phytogeographia Aegaea and the Flora Hellenica database. Ann Nat Hist Mus Wien Ser B Bot Zool:279–289
18. Rechinger KH (1943) Flora Aegaea. Flora der Inseln und Halbinseln des ägäischen Meeres. Akad Wiss Wien Math-Naturwiss Kl Denkschr 105(1)
19. Turrill WB (1929) The plant-life of the Balkan Peninsula: a phytogeographical study, vol 1. Clarendon Press, Oxford
20. Watson GE (1964) Ecology and evolution of passerine birds on the islands of the Aegean Sea. PhD. Thesis. Yale University, New Haven, CT
21. Giokas S (2000) Congruence and conflict in Albinaria (Gastropoda, Clausiliidae). A review of morphological and molecular phylogenetic approaches. Belg J Zool 130:93–100
22. Kougioumoutzis K, Valli AT, Georgopoulou E, Simaiakis SM, Triantis KA, Trigas P (2016) Network biogeography of a complex island system: the Aegean Archipelago revisited. J Biogeogr. doi:10.1111/jbi.12920
23. Poulakakis N, Kapli P, Lymberakis P, Trichas A, Vardinogiannis K, Sfendourakis S, Mylonas M (2014) A review of phylogeographic analyses of animal taxa from the Aegean and surrounding regions. J Zoolog Syst Evol Res 53:18–32
24. Sfenthourakis S, Triantis KA (2017) The Aegean archipelago: a natural laboratory of evolution, ecology and civilisations. J Biol Res Thessalon 24(1):1–13
25. Ganiatsas K (1967) Phytogeographia, Thessaloniki
26. Audiso P, De Biase A, Jäch MA (1996) Greek species of the genus Hydraena (subgenus Haenydra Rey) (Coleoptera, Hydraenidae). Aquat Insects 18(2):65–90
27. Dimopoulos P, Raus T, Bergmeier E, Constantinidis T, Iatrou G, Kokkini S, Strid A, Tzanoudakis D (2013) Vascular plants of Greece: an annotated checklist. Botanischer Garten und Botanisches Museum/Hellenic Botanical Society, Berlin/Athens
28. Phitos D, Constantinidis T, Kamari G (2009) The red data book of rare and threatened plants of Greece2nd edn. Hellenic Botanical Society, Patras
29. Legakis A, Pafilis P, Parmakelis A (2015) Editorial. J Nat Hist 49:2–8

30. Daget J, Economidis PS (1975) Richesse spécifique de l'ichtyofaune de Macédoine orientale et de Thrace occidentale (Grèce). Bull Mus Natl Hist Nat Paris 346:81–84
31. Economou AN, Giakoumi S, Vardakas L, Barbieri R, Stoumboudi M, Zogaris S (2007) The freshwater ichthyofauna of Greece: an update based on a hydrographic basin survey. Mediterr Mar Sci 8:91–168
32. Bianco PG (1990) Potential role of the paleohistory of the Mediterranean and Paratethys basins on the early dispersal of Euro-Mediterranean freshwater fish. Ichthyol Explor Freshw 1:167–184
33. Economidis PS, Bănărescu PM (1991) The distribution and origins of freshwater fishes in the Balkan peninsula, especially in Greece. Int Rev Gesamten Hydrobiol 76:257–283
34. Maurakis EG, Economidis PS (2001) Reconstructing biogeographic relationships of river drainages in Peloponnessuos, Greece using distributions of freshwater cyprinid fishes. Bios 6:125–132
35. Reyjol Y, Hugueny B, Pont D, Bianco PG, Beier U, Caiola N, Casals F, Cowx I, Economou AN, Ferreira T, Haidvogl G, Noble R, de Sostoa A, Vigneron T, Virbickas T (2007) Patterns in species richness and endemism of European freshwater fish. Glob Ecol Biogeogr 16:65–75
36. Zogaris S, Economou A, Dimopoulos P (2009) Ecoregions in the Southern Balkans: should they be revised? Environ Manag 43:682–697
37. Zogaris S, Legakis A, Rejyol Y, Chatzinikolaou Y, Dimopoulos P, Economou AN (2009) Freshwater biogeographic units in Greece: where to draw the lines? 11th International Congress on the Zoogeography and Ecology of Greece and Adjacent Regions-ICZEGAR, proceedings, 21–25 September 2009, Herakleion, p 204
38. Mazaris AD, Moustaka-Gouni M, Michaloudi E, Bobori DC (2010) Biogeographic patterns of freshwater micro and macroorganisms: a comparison between phytoplankton, zooplankton and fish in the eastern Mediterranean. J Biogeogr 37(7):1341–1351
39. Economou AN, Zogaris S, Vardakas L, Koutsikos N, Chatzinikolaou Y, Kommatas D, Kapakos Y, Giakoumi S, Oikonomou E, Tachos V (2016) Developing policy-relevant river fish monitoring in Greece: insights from a nation-wide survey. Mediterr Mar Sci 171:302–322
40. Illies J (1967, 1978) Limnofauna Europaea. Gustav Fisher Verlag, Stuttgart
41. Tierno de Figueroa JM, López-Rodríguez MJ, Fenoglio S, Sánchez-Castillo P, Fochetti R (2013) Freshwater biodiversity in the rivers of the Mediterranean Basin. Hydrobiologia 719 (1):37–186
42. Csabai Z, Soos N, Berchi GM, Cianferoni F, Boda P, Mora A (2017) Aquatic and semiaquatic Heteroptera (Nepomorpha and Gerromorpha) fauna of Greek holiday islands (Rhodes, Crete and Corfu) with first records of three species from Europe and Greece. Zootaxa 4231(1):51–69
43. Bailey RG (2002) Ecoregion-based design for sustainability. Springer, New York, NY
44. Loveland TR, Merchant JM (2004) Ecoregions and ecoregionalization: geographical and ecological perspectives. Environ Manag 34:(1): S1–(1):S13
45. Dinerstein E, Olson D, Joshi A, Vynne C, Burgess ND, Wikramanayake E, Hahn N, Palminteri S, Hedao P, Noss R, Hansen M, et al. (2017) An ecoregion-based approach to protecting half the terrestrial realm. Bioscience. doi:10.1093/biosci/bix014
46. Bohn U, Gollub G, Hettwer C, Neuhäuslová Z, Raus T, Schlüter H, Weber H (2004) Map of the natural vegetation of Europe, scale 1:2.500.000, interactive CD-ROM – explanatory text, legend, maps. Landwirtschaftsverlag, Münster
47. Mavrommatis GN (1980) The bioclimate of Greece: relationships between climate and vegetation – bioclimatic maps. Institute for Forestry Research, Athens. (in Greek)
48. Triantis KA, Mylonas M (2009) Greek islands, biology. In: Gillespie R, Glague DA (eds) Encyclopedia of islands. University of California Press, Los Angeles, CA
49. Collier RELI, Leeder MR, Jackson JA (1995) Quaternary drainage development, sediment fluxes and extensional tectonics in Greece. In: Lewin J, Macklin MG, Woodward JC (eds) Mediterranean Quaternary river environments. A.A. Balkema, Rotterdam, pp 31–44

50. Higgins MD, Higgins R (1996) A geological companion to Greece and the Aegean. Cornell University Press, New York, NY
51. Skoulikidis N, Economou AN, Gritzalis KC, Zogaris S (2009) Rivers of the Balkans. In: Tockner K, Uehlinger U, Robinson CT (eds) Rivers of Europe. Elsevier Academic Press, Amsterdam
52. Valakos S, Pafilis P, Sotiropoulos K, Lymberakis P, Maragou P, Foufopoulos J (2008) Amphibians and reptiles of Greece. Chimaira Editions, Frankfurt-am-Main
53. Allen DH (2001) Mediterranean ecogeography. Pearson Education, London
54. Tselepedakis IG, Theoharatos GA (1989) A bioclimatic classification of the Greek area. Theor Appl Climatol 40:147–153
55. Baltas EA (2008) Climatic conditions and availability of water resources in Greece. Int J Water Resour D 24(4):635–649
56. Markonis Y, Batelis SC, Dimakos Y, Moschou E, Koutsoyiannis D (2016) Temporal and spatial variability of rainfall over Greece. Theor Appl Climatol. doi:10.1007/s00704-016-1878-7
57. Skoulikidis NT, Vardakas L, Karaouzas I, Economou AN, Dimitriou E, Zogaris S (2011) Assessing water stress in a Mediterranean lotic system: insights from an artificially intermittent river in Greece. Aquat Sci 73:581–597
58. Barbieri R, Zogaris S, Kalogianni E, Stoumboudi M, Chatzinikolaou Y, Giakoumi S, Kapakos Y, Kommatas D, Koutsikos N, Tachos V, Vardakas L, Economou AN (2015) Freshwater fishes and lampreys of Greece: an annotated checklist. Monographs on marine sciences no. 8. Hellenic Center for Marine Research, Athens
59. Vanhove MPM, Economou AN, Zogaris S, Giakoumi S, Zanella D, Volckaert FA, Huyse T (2013) The Gyrodactylus (Monogenea, Gyrodactylidae) parasite fauna of freshwater sand gobies (Teleostei, Gobioidei) in their centre of endemism, with description of seven new species. Parasitol Res 113:653–668
60. Strid A, Tan K (1997) Flora Hellenica, vol I. University of Copenhagen, Koeltz
61. Georghiou K, Delipetrou P (2010) Patterns and traits of the endemic plants of Greece. Bot J Linn Soci 162:130–422
62. Trigas P, Panitsa M, Tsiftsis M (2013) Elevational gradient of vascular plant species richness and endemism in Crete – the effect of post-isolation mountain uplift on a continental island system. PLoS One 8:e59425
63. Blaženčić J, Stevanović B, Blaženčić Ž, Stevanović V (2006) Red data list of charophytes in the Balkans. Biol Conserv 15:3445–3457
64. Sarika M, Dimopoulos P, Yannitsaros A (2005) Contribution to the knowledge of the wetland flora and vegetation of Amvrakikos Gulf, W Greece. Willdenowia 35:69–85
65. Catsadorakis G, Paragamian K (2007) Inventory of the wetlands of the Aegean Islands: identity, ecological status and threats. WWF Greece, Athens. (in Greek)
66. Koumpli-Sovantzi L (1997) The charophyte flora of Greece. I Fl Medit 7:173–179
67. Manolaki P, Papastergiadou E (2013) The impact of environmental factors on the distribution pattern of aquatic macrophytes in a middle-sized Mediterranean stream. Aquat Bot 104:34–46
68. Douda J, Boublík K, Slezák M, Biurrun I, Nociar J, Havrdová A, Doudová J, Aćić S, Brisse H, Brunet J, Chytrý M, Claessens H, Csiky J, Didukh Y, Dimopoulos P, Dullinger S, FitzPatrick U, Guisan A, Horchler PJ, Hrivnak R, Jandt U, Kazcki Z, Kevey B, Landucci F, Lecomte H, Lenoir J, Paal J, Paternoster D, Pauli H, Pielech R, Rodwell JS, Roelandt B, Svenning J-C, Sibık J, Silc U, Skvorc U, Tsiripidis I, Tzonev RT, Wohlgemuth T, Zimmermann NE (2016) Vegetation classification and biogeography of European floodplain forests and alder carrs. Appl Veg Sci 19(1):147–163
69. Dimopoulos P, Zogaris S (2008) Vegetation and flora of riparian zones. In: Arizpe D, Mendes A, Rabaca J (eds) Sustainable riparian zones: a management guide. Generalitat Valenciana, Valencia, pp 66–82

70. Zogaris S, Chatzinikolaou Y, Dimopoulos P (2008) Riparian woodland flora in upland rivers of Western Greece. Mediterr Mar Sci 9(2):87–103
71. Coughlan NE, Kelly TC, Davenport J, Jansen MA (2017) Up, up and away: bird-mediated ectozoochorous dispersal between aquatic environments. Freshw Biol. doi:10.1111/fwb.12894
72. Leeuwen CH, Velde G, Groenendael JM, Klaassen M (2012) Gut travellers: internal dispersal of aquatic organisms by waterfowl. J Biogeogr 39(11):2031–2040
73. Lopes-Lima M, Sousa R, Geist J, Aldridge D, Araujo R, Bergengren J, Bespalaja Y, Bódis E, Burlakova L, Van Damme D, Douda K, Froufe E, Georgiev D, Gumpinger C, Karatayev A, Kebapçi Ü, Killeen I, Lajtner J, Larsen B, Lauceri R, Legakis A, Lois S, Lundberg S, Moorkens E, Motte G, Nagel K, Ondina P, Outeiro A, Paunovic M, Prié V, Von Proschwitz T, Riccardi N, Rudzīte M, Rudzītis M, Scheder C, Seddon M, Şereflişan H, Simić V, Sokolova S, Stoeckl K, Taskinen J, Teixeira A, Thielen F, Trichkova T, Varandas S, Vicentini H, Zajac K, Zajac T, Zogaris S (2015) Conservation status of freshwater mussels in Europe: state of the art and future challenges. Biol Rev 92:572–607
74. Harzhauser M, Piller WE, Steininger FF (2002) Circum-Mediterranean Oligo-Miocene biogeographic evolution – the gastropods' point of view. Palaeogeogr Palaeocl 183:103–133
75. Gloer P, Fralnioski A, Pesic V (2010) The Bithynidae of Greece (Gasteropoda: Bithynidae). J Conchol 40(2):179–187
76. Radea C (2011) A new species of hydrobiid snails (Mollusca, Gastropoda, Hydrobiidae) from central Greece. ZooKeys 138:53–64
77. Radea C, Parmakelis A, Papadogiannis V, Charou D, Triantis KA (2013) The hydrobioid freshwater gastropods (Caenogastropoda, Truncatelloidea) of Greece: new records, taxonomic re-assessments using DNA sequence data and an update of the IUCN red list categories. ZooKeys 350:1–20
78. Jäch MA, Avtzis ND (1998) Check list of the Dryopidae, Elmidae and Hydraenidae (Coleoptera) of Greece. Entomol Hell 12:13–17
79. Trichas A (1996) Ecology and biogeography of the ground Coleoptera in the south Aegean area. Doctorate dissertation. University of Crete, Heraklion
80. Legakis A, Maragou P (2009) Greek red data book of threatened fauna. Hellenic Zoological Society, Athens
81. Malicky H (1979) The caddisflies (Insecta, Trichoptera) of the Ionian Islands. Biol Gallo-Hell 8:141–145
82. Sfenthourakis S, Legakis A (2001) Hotspots of endemic terrestrial invertebrates in southern Greece. Biodivers Conserv 10:1387–1417
83. Welter-Schultes FW (2000) The paleogeography of late Neogene central Crete inferred from the sedimentary record combined with Albinaria land snail biogeography. Palaeogeogr Palaeoclimatol Palaeoecol 157:27–44
84. Solà E, Sluys R, Gritzalis K, Riutort M (2013) Fluvial basin history in the northeastern Mediterranean region underlies dispersal and speciation patterns in the genus Dugesia (Platyhelminthes, Tricladida, Dugesiidae). Mol Phylogenet Evol 66:877–888
85. Christodoulou M, Anastasiadou C, Jugovic J, Tzomos T (2016) Freshwater shrimps (Atyidae, Palaemonidae, Typhlocarididae) in the broader Mediterranean region: distribution, life strategies, threats, conservation challenges and taxonomic issues. In: Kawai T, Cumberlidge N (eds) A global overview of the conservation of freshwater decapod crustaceans. Springer, Cham, pp 199–236
86. Hupało K, Mamos T, Olszewska W, Wysocka A, Karaouzas I, Plaiti W, Michał Grabowski M (2015) Diversity and origin of freshwater gammarids from Crete and Peloponnese – preliminary results. In: Poulakakis N et al. (eds) Abstracts of the International Congress on the Zoogeography and Ecology of Greece and Adjacent Regions, 13th ICZEGAR, 7–11 October 2015, Irakleio, Crete, Greece. Hellenic Zoological Society, Irakleio

87. Karaouzas I, Gritzalis KC (2006) Local and regional factors determining aquatic and semi-aquatic bug (Heteroptera) assemblages in rivers and streams of Greece. Hydrobiologia 573:199–212
88. Karaouzas I, İncekara Ü (2011) First record of the genus Berosus (Coleoptera: Hydrophilidae) in Crete island, Greece. Entomol Hell 20:69–74
89. Karaouzas I (2013) The larvae of the Aegean endemic caddisflies Hydropsyche debirasi Malicky 1974 and Hydropsyche kleobis Malicky 2001 (Trichoptera: Hydropsychidae) with notes on their ecology and zoogeography. Zootaxa 3636(4):472–477
90. Płóciennik M, Karaouzas I (2014) The Chironomidae (Diptera) fauna of Greece: zoogeographical distributions and patterns, taxalist and new records. Ann Limnol-Int J Lim 50:19–34
91. Bauernfeind E (2003) The mayflies of Greece (Insecta: Ephemeroptera) – a provisional check-list. Research update on Ephemeroptera and Plecoptera. University of Perugia, Perugia, pp 99–107
92. Karaouzas I, Adriopoulou A, Kouvarda T, Muranyi D (2016) An annotated checklist of the Greek stonefly fauna (Insecta: Plecoptera). Zootaxa 4111:301–333
93. Argyroudi A, Chatzinikolaou Y, Poirazidis K, Lazaridou M (2009) Do intermittent and ephemeral Mediterranean rivers belong to the same river type? Aquat Ecol 43:465–476
94. Skoulikidis NT, Karaouzas I, Gritzalis KC (2008) Identifying key environmental variables structuring benthic fauna for establishing a biotic typology for Greek running waters. Limnologica 39:55–66
95. Radea C, Parmakelis A, Giokas S (2016) Myrtoessa hyas, a new valvatiform genus and a new species of the Hydrobiidae (Caenogastropoda, Truncatelloidea) from Greece. ZooKeys 640:1–18
96. Olden JD, Kennard MJ, Leprieur F, Tedesco PA, Winemiller KO, García-Berthou E (2010) Conservation biogeography of freshwater fishes: recent progress and future challenges. Divers Distrib 16(3):496–513
97. Dias MS, Oberdorff T, Hugueny B, Leprieur F, Jézéquel C, Cornu JF, Brosse S, Grenouillet G, Tedesco PA (2014) Global imprint of historical connectivity on freshwater fish biodiversity. Ecol Lett 17(9):130–1140
98. Geiger MF, Herder F, Monaghan MT, Almada V, Barbieri R, Bariche M, Berrebi P, Bohlen J, Casal-Lopez M, Delmastro GB, Denys GPJ, Dettai A, Doadrio I, Kalogianni E, Kärst H, Kottelat M, Kovačić M, Laporte M, Lorenzoni M, Marčić M, Özuluğ M, Perdices A, Perea S, Persat H, Porcelotti S, Puzzi C, Robalo J, Šanda R, Schneider M, Šlechtová V, Stoumboudi M, Walter S, Freyhof J (2014) Spatial heterogeneity in the Mediterranean biodiversity hotspot affects barcoding accuracy of its freshwater fishes. Mol Ecol Resour 14(6):1210–1221
99. Tsigenopoulos CS, Berrebi P (2000) Molecular phylogeny of North Mediterranean freshwater barbs (genus Barbus: Cyprinidae) inferred from cytochrome b sequences: biogeographic and systematic implications. Mol Phylogenet Evol 14(2):165–179
100. Petit G (1930) Remarques sur la distribution geographique des poisons d'eau douce des Balkans. C R Soc Seances Soc Biol Fil 59:77–82
101. Smith KG, Darwall WRT (eds) (2006) The status and distribution of freshwater fish endemic to the Mediterranean Basin. IUCN, Gland
102. Froese R, Pauly D (eds) (2000) FishBase 2000: concepts, design and data sources. ICLARM, Los Baños, Laguna
103. Šanda R, Vukic J, Choleva L, Krízek J, Sedivá A, Shumka S, Wilson IF (2008) Distribution of loach fishes (Cobitidae, Nemacheilidae) in Albania, with genetic analysis of populations of Cobitis ohridana. Folia Zool 57(1/2):42
104. Vukić J, Ulqini D, Šanda R (2017) Occurrence of Knipowitschia goerneri Ahnelt, 1991 (Gobiidae) in southern Albania confirmed with molecular tools. J Appl Ichthyol 33(2):284–290
105. Bobori DC, Economidis PS (2006) Freshwater fishes of Greece: their biodiversity, fisheries and habitats. Aquat Ecosyst Health Manage 9:407–418

106. Pafilis P, Valakos SD (2012) Amphibians and reptiles of Greece: guide to identification. Patakis, Athens. (in Greek)
107. Sotiropoulos, K. (2004) Study on the variation of the genus Triturus (Amphibia, Urodela) in Greece, based on biochemical, morphological and ecological criteria. Doctoral Thesis. National and Kapodistrian University of Athens, Dept. of Biology, Section of Zoology and Marine Biology, Athens
108. Foufopoulos J, Ives AR (1999) Reptile extinctions on land-bridge islands: life-history attributes and vulnerability to extinction. Am Nat 153:1–25
109. Handrinos G, Akriotis T (1997) Birds of Greece. Christopher Helm, London
110. Simaiakis SM, Dretakis M, Barboutis C, Katritis T, Portolou D, Xirouchakis S (2012) Breeding land birds across the Greek islands: a biogeographic study with emphasis on faunal similarity, species–area relationships and nestedness. J Ornithol 153(3):849–860
111. Catsadorakis G, Onmuş O, Bugariu S, Gül O, Hatzilacou D, Hatzofe O, Malakou M, Michev T, Naziridis T, Nikolaou H, Rudenko A (2015) Current status of the Dalmatian pelican and the great white pelican populations of the Black Sea/Mediterranean flyway. Endanger Species Res 27(2):119–130
112. Green AJ, Elmberg J (2014) Ecosystem services provided by waterbirds. Biol Rev 89(1):105–122
113. Stuart AJ, Lister AM (2007) Patterns of Late Quaternary megafaunal extinctions in Europe and northern Asia. Cour Forsch Inst Senckenberg 259:287–297
114. Van der Geer A, Dermitzakis M, De Vos J (2006) Crete before the Cretans: the reign of dwarfs. Pharos 13:121–132
115. De Vos J, van den Hoek Ostende LW, van den Bergh GD (2007) Patterns in insular evolution of mammals: a key to island palaeogeography. In: Renema W (ed) Biogeography, time, and place: distributions, barriers, and islands. Springer, Dordrecht
116. Blondel J (2008) On humans and wildlife in Mediterranean islands. J Biogeogr 35:509–518
117. Sfikas G (1985) Greek nature through the ages. Mikri Vivliothiki, Athens. (in Greek)
118. Voultsiadou E, Tatolas A (2005) The fauna of Greece and adjacent areas in the Age of Homer: evidence from the first written documents of Greek literature. J Biogeogr 32 (11):1875–1882
119. Masseti M (2012) Atlas of terrestrial mammals of the Ionian and Aegean islands. Walter de Gruyter, Berlin
120. Karandinos M, Paraschi L (1992) The red data book of threatened vertebrates of Greece. Hellenic Zoological Society and Hellenic Ornithological Society, Athens
121. Halley D, Rosell F, Saveljev A (2012) Population and distribution of Eurasian beaver (Castor fiber). Balt For 18:168–175
122. Catsadorakis G, Källander H (eds) (2010) The Dadia-Lefkimi-Soufli Forest National Park, Greece: biodiversity, management and conservation. WWF Greece, Athens
123. Perissoratis C, Conispoliatis N (2003) The impacts of sea-level changes during latest Pleistocene and Holocene times on the morphology of the Ionian and Aegean Seas (SE Alpine Europe). Mar Geol 196:145–156
124. Oikonomou A, Leprieur F, Leonardos ID (2014) Biogeography of freshwater fishes of the Balkan Peninsula. Hydrobiologia 738(1):205–220
125. Atalay I (2002) Ecoregions of Turkey. Ministry of Forestry, Publication n. 163, Ankara
126. Lykousis V, Karageorgis AP, Chronis GT (2005) Delta progradation and sediment fluxes since the last glacial in the Thermaikos Gulf and the Sporades Basin, NW Aegean Sea, Greece. Mar Geol 222–223:381–397
127. Kottelat M, Freyhof J (2007) Handbook of European freshwater fishes. Kottelat, Cornol and Freyhof, Berlin
128. Bianco PG (1986) The zoogeographic units of Italy and Western Balkans based on cyprinid species ranges (Pisces). Biol Gallo-Hell 12:291–299
129. Stierandová S, Šanda R, Vasil'eva ED, Zogaris S, Shumka S, Halačka K, Vetešník L, Nowak M, Stefanov T, Koščo J, Mendel J (2016) A multilocus assessment of nuclear and

mitochondrial sequence data elucidates phylogenetic relationships among European spirlins (Alburnoides, Cyprinidae). Mol Phylogenet Evol 94:479–491
130. Albrecht C, Wilke T (2008) Ancient Lake Ohrid: biodiversity and evolution. Hydrobiologia 615(1):103–140
131. Stephanidis A (1939) Freshwater fish of Attico-Beotia. Bull Phys Sci Fasc 50–51:49–60. (in Greek)
132. Lykousis V (2009) Sea-level changes and shelf break prograding sequences during the last 400ka in the Aegean margins: subsidence rates and palaeogeographic implications. Cont Shelf Res 29(16):2037–2044
133. Malicky H (1987) The miraculous island of Serifos-one possible key to understand the evolution of Mediterranean stream ecosystems. Biol Gallo-Hell 13:43–46
134. Welter-Schultes FW, Williams MR (1999) History, island area and habitat availability determine land snail species richness of Aegean islands. J Biogeogr 26(2):239–249
135. Schmitt T (2007) Molecular biogeography of Europe: Pleistocene cycles and postglacial trends. Front Zool 4:11. doi:10.1186/1742-9994-4-11
136. Radea C, Louvrou I, Bakolitsas K, Economou-Amilli A (2017) Local endemic and threatened freshwater hydrobiids of Western Greece: elucidation of anatomy and new records. Fol Malac 25(1):3–13
137. Froufe E, Prié V, Faria J, Ghamizi M, Gonçalves DV, Gürlek ME, Karaouzas I, Kebapçi Ü, Şereflişan U, Sobral C, Sousa R, Teixeira A, Varandas S, Zogaris S, Lopes-Lima M (2016) Phylogeny, phylogeography, and evolution in the Mediterranean region: news from a freshwater mussel (Potomida, Unionida). Mol Phylogenet Evol 100:322–332
138. Christodoulou M, Antoniou A, Magoulas A, Koukouras A (2012) Revision of the freshwater genus Atyaephyra (Crustacea, Decapoda, Atyidae) based on morphological and molecular data. ZooKeys 229:53–110. doi:10.3897/zookeys.229.3919
139. Vukić J, Kovačić M, Zogaris S, Šanda R (2016) Rediscovery of Knipowitschia goerneri (Actinopterygii: Gobiidae) and its phylogenetic relationships with other northern Mediterranean Knipowitschia species. Ichthyol Explor Fres 26:363–372
140. Iliadou E, Kallimanis AS, Dimopoulos P, Panitsa M (2014) Comparing the two Greek archipelagos plant species diversity and endemism patterns highlight the importance of isolation and precipitation as biodiversity drivers. J Biol Res Thessalon 21:16
141. Vanhove MPM, Kovačić M, Zogaris S (2016) A distinct island population of threatened freshwater fish: to split or lump? Hydrobiologia 777:79–93
142. Rackham O, Moody J (1996) The making of the Cretan landscape. Manchester University Press, Manchester
143. Raus T (1991) Asia or Europe? – the phytogeographical position of the Karpathos archipelago (SE Aegean, Greece). Fl Veg Mundi 9:301–310
144. Creutzburg N (1963) Paleogeographic evolution of Crete from Miocene till our days. Cret Ann 15:336–342
145. Dermitzakis MD, Papanikolaou D (1981) Paleogeography and geodynamics of the Aegean region during the Neogene. Ann Geol Pays Hell 4:245–289
146. Legakis A (1987) Primary zoogeographic analysis of Crete. Biol Gallo-Hell 13:133–136
147. Markonis Y, Angelakis AN, Christy J, Koutsoyiannis D (2016) Climatic variability and the evolution of water technologies in Crete, Hellas. Water Hist 8:137–157. doi:10.1007/s12685-016-0159-9
148. Soukissian T, Hatzinaki M, Korres G, Papadopoulos A, Kallos G, Anadranistakis E (2007) Wind and wave atlas of the Hellenic seas. Hellenic Centre for Marine Research, Anavissos
149. Barbieri R, Stoumboudi MT, Giakoumi S, Kapakos Y, Skoulikidis N, Kalogianni E (2013) The fish fauna of the inland waters of Crete Island. 15th Panhellenic Ichthyological Congress, Thessaloniki, 10–13 October 2013. (in Greek with English abstract)
150. Legakis A, Kypriotakis Z (1994) A biogeographic analysis of the island of Crete, Greece. J Biogeogr 21:441–445

151. Stoumboudi MT, Kottelat M, Barbieri R (2006) The fishes of the inland waters of Lesbos Island, Greece. Icthyol Explor Freshw 17(2):129–146
152. Bianco PG, Ahnelt H, Economidis PS (1996) The freshwater fish from eastern and large Mediterranean islands with comments on their safety status. Acta Univ Carol 40:45–60
153. Davis PH (1965) Flora of Turkey and the east Aegean Islands, vol 1. Edinburgh University Press, Edinburgh
154. Fattorini S (2002) Biogeography of the tenebrionid beetles (Coleoptera, Tenebrionidae) on the Aegean Islands (Greece). J Biogeogr 29:49–67
155. Hausdorf B, Hennig C (2005) The influence of recent geography, palaeogeography and climate on the composition of the fauna of the central Aegean Islands. Biol J Linn Soc 84:785–795
156. Geldiay R, Balik S (1999) Freshwater fishes of Turkey. Ege Universitesi Basimevi, Bornova-Izmir. (in Turkish)
157. Greuter W (1979) The flora and phytogeography of Kastellorizo (Dhodhekanisos, Greece). 1. An annotated catalogue of the vascular plant taxa. Willdenowia 8(3):531–611
158. Constantinidis T (2013) The flora of the Kastellorizo island group (East Aegean Islands, Greece): new records and comments. Fl Medit 23:69–86
159. Koutsikos N, Zogaris S, Vardakas L, Tachos V, Kalogianni E, Šanda R, Chatzinikolaou Y, Giakoumi S, Economidis PS, Economou AN (2012) Recent contributions to the distribution of the freshwater ichthyofauna in Greece. Mediterr Mar Sci 13(2):268–277
160. Higgins JV, Bryer MT, Khoury ML, Fitzhugh TW (2005) A freshwater classification approach for biodiversity conservation planning. Conserv Biol 19:432–445
161. Forman RT (1995) Land mosaics: the ecology of landscapes and regions. Island Press, Washington DC
162. Blackmore S (1996) Knowing the Earth's biodiversity: challenges for the infrastructure of systematic biology. Science 274(5284):63
163. Darwall WRT, Carrizosa S, Numa C, Barrios V, Freyhof J, Smith K (2014) Freshwater key biodiversity areas in the Mediterranean Basin hotspot: informing species conservation and development planning in freshwater ecosystems. IUCN, Cambridge
164. Zenetos A, Pancucci-Papadopoulou MA, Zogaris S, Papastergiadou E, Vardakas AL, Aligizaki K, Economou AN (2009) Aquatic alien species in Greece (2009): tracking sources, patterns and effects on the ecosystem. J Biol Res Thessalon 12:135–172
165. Vanhove MPM, Economou AN, Zogaris S, Giakoumi S, Kalogianni E, Volckaert FAM, Huyse T (2011) Phylogenetics and biogeography of Balkan 'sand gobies' (Teleostei: Gobiidae): vulnerable species in need of taxonomic revision. Biol J Linn Soc 105:73–91
166. Hartshorne R (1958) The concept of geography as a science of space, from Kant and Humboldt to Hettner. Ann Assoc Am Geogr 48(2):97–108
167. McMahon G, Wiken EB, Gauthier DA (2004) Towards a scientifically rigorous basis for developing mapped ecological regions. Environ Manag 34:111–124
168. Vlami V, Kokkoris IP, Zogaris S, Cartalis C, Kehayias G, Dimopoulos P (2017) Cultural landscapes and attributes of "culturalness" in protected areas: an exploratory assessment in Greece. Sci Total Environ 595:229–243
169. Kyriazi P, Kornilios P, Nagy ZT, Poulakakis N, Kumlutaş Y, Ilgaz C, Avcı A, Göçmen B, Lymberakis P (2013) Comparative phylogeography reveals distinct colonization patterns of Cretan snakes. J Biogeogr 40:1143–1155
170. Skoulikidis N, Lampou A, Karaouzas I, Gritzalis K, Zogaris S (2014) Stream ecological assessment on an Aegean island: insights from an exploratory application on Samothraki (Greece). Fresen Environ Bull 23(5):1173–1182
171. Malicky H (2005) Die Köcherfliegen Griechenlands. Denisia 17:1–240

The State and Origin of River Water Composition in Greece

Nikos Skoulikidis

Abstract This chapter provides an overview on the hydrogeochemical and pollution characteristics of Greek rivers and attempts to interpret the origin and spatio-temporal variability of their composition as it emerges from various natural factors and processes and human interference. Despite the highly variable physicogeographic and geological conditions of the country, river basins may be hydrogeochemically classified into three main geographical zones with distinct geological, climatic and hydrological features. River hydrogeochemical properties mainly depend on geochemical, hydromorphological and climatic factors. Catchment geology directly controls solute concentrations and major ion portions and influences hydrological and hydrogeological factors. The latter indirectly control water temperature and solute concentrations, as well as pH and carbonate equilibrium together with biological activity. In certain river basins, anthropogenic pressures (i.e. inadequately treated municipal wastes, agrochemicals, agro-industrial and mining effluents) affect aquatic quality, whereas water resources management (i.e. water overexploitation for irrigation and dam operation) alters the hydrological regime, thus indirectly influencing solute concentration. In general, rivers located in western Greece as well as mountainous rivers and streams range from "pristine" to satisfactory conditions. On the contrary, lowland sections of large rivers are at a greater risk due to a variety of pressures, such as agriculture, agro-industry, mining, (illegal) building, and tourism. Despite the great number of internationally important sites in river basins and, the recent, major efforts made in implementing the WFD, Greek rivers are still threatened from insufficient implementation of environmental legislation and ad-hoc management practices. The economic crisis may set environmental conservation at the expense of economic development and/or

N. Skoulikidis (✉)
Hellenic Centre for Marine Research, Institute of Marine Biological Resources and Inland Waters, 46.5 km Athens-Sounio, P.O. Box 712, 19013 Anavissos, Greece
e-mail: nskoul@hcmr.gr

N. Skoulikidis et al. (eds.), *The Rivers of Greece: Evolution, Current Status and Perspectives*, Hdb Env Chem (2018) 59: 97–128, DOI 10.1007/698_2016_468,

change socio-economical attitudes. thus pushing environmental conservation forward.

Keywords Aquatic quality, Biogeochemistry, Climate, Geology, Pollution

Contents

1 Introduction

Greece is a small mountainous country with marked relief variability within small spatial scales so that mountain peaks lie close to plains and the long coastline, complex geological structures, a palette of microclimates, and a variety of vegetation cover and aquatic ecosystems hosting rich biodiversity. As a consequence of its geologically young morphology with mountainous relief (65% of the country lies over 200 m), high altitudinal differences and steep slopes, prolonged dry summer period (especially in semi-arid areas), and the high amount of impermeable rocks (30% of the country's surface), Greece is characterised by numerous, highly fragmented, small to medium sized mountainous rivers and streams, with flushy flow and sediment regimes, running through steep narrow valleys and descending abruptly to the coast. In semi-arid landscapes and, particularly in islands, non-perennial streams with intermittent to episodic flow regimes dominate. Large lowland areas are scarcely diffused within prevailing thrust belts and related rift valleys. These flat areas are drained by medium and large low-gradient perennial rivers with runoff of several km^3/year, which frequently form extensive flood and deltaic plains.

Figure 1 presents the main Greek rivers, discussed in the present chapter and Table 1 presents their physicogeographical features. There are 765 recorded rivers in total, of which 45 are registered as permanently flowing according to the Ministry for Development [2], with ten of them exceeding 1 km^3/a [3]. From the latter river basins, the interregional ones, shared between Greece/Bulgaria (Evros and Strymon) and Greece/FYR Macedonia (Axios), together with Pinios, situated in central Greece, are considered very large (>10,000 km^2). There are fifteen large river basins (1,000–10,000 km^2), twenty medium sized (100–1,000 km^2) and about ten small-sized (10–100 km^2) ones, with perennial flow, the majority of which drain the

Fig. 1 Map showing the geographic (climatic-geological-hydrochemical) zones and the main Greek river catchments discussed in this chapter

western part of the country. The total mean annual surface runoff of the Greek Peninsula is estimated to be around 54 km^3 [2], thereby comprising between 16 and 27% of the European river runoff in the Mediterranean (~330 km^3/a, [4]; ~200 km^3/a, [5]); 12.6% of the total Mediterranean runoff (430 km^3/a, [4]); 2% of the total European runoff (2,770 km^3/a, [6]); and 0.13% of the world's river runoff (39,394 km^3/a, [6]). These numbers may be considered high since the surface area of Greece comprises only 7% of the European Mediterranean area,[1] and 1.3% of the European area. Despite high total discharge outputs, the spatial and temporal distribution of surface runoff is highly uneven due to climatic and geological variability. In addition, the main portion of river basin networks,

[1]Including Spain, France, Italy, Slovenia, Croatia, Bosnia-Herzegovina, Montenegro, Albania, FYROM, Bulgaria and Greece.

Table 1 Physicogeographic characteristics of major Greek rivers

	Hydromorphological characteristics							Geology		Land use[a]					
	E[b]	CA[c]	L[c]	P[b]	T[b]	Q[d]	SQ			Ur	Gr	Cr	Shrub	For	OW
	m	km^2	km	mm	°C	km^3	$l/s*km^2$	si	carb	% of catchment					
Zone 3 (rivers listed from north to south)															
Aoos	845	6,651	249	1,119	11.0	4.51	21.50		+	2.4	11.0	18.5	19.3	48.4	0.5
Kalamas	497	1,899	137	1,174	13.2	1.22	20.44			1.6	10.8	27.0	19.2	40.4	1.0
Arachthos	829	2,443	149	1,005	11.5	1.22	15.81		+	3.4	11.3	22.1	24.1	36.7	2.3
Louros	397	931	88	1,036	14.4	0.38	13.06		+	3.2	9.6	38.5	29.1	15.3	4.2
Acheloos	812	5,688	257	924	12.3	2.32	12.92		+	1.5	5.8	18.3	17.4	52.1	4.8
Krathis	1,051	155	27	869	11.4	0.32	64.87		+	1.2	3.8	17.0	19.4	58.5	0.0
Alpheios	679	3,501	145	851	13.6	1.71	15.53		+	1.7	7.0	38.6	19.7	32.7	0.3
Evrotas[e]	632	1,738	92	772	14.0	0.76	13.93		+	1.0	8.1	34.6	28.9	27.4	0.2
Anapodaris[f]	444	517	43	739	16.7	0.01	0.54		+	0.6	15.9	70.2	12.3	0.9	0.1
Zone 2 (rivers listed from north to south)															
Soulou	862	966	42	616	10.3	0.37	12.13	+		6.7	7.5	41.4	25.2	19.1	0.0
Aliakmonas	817	6,583	322	702	10.8	2.34	11.27	+		1.7	9.2	38.8	8.7	39.6	2.1
Pinios	417	10,845	376	633	14.1	2.38	6.97	+		2.8	10.1	52.2	17.0	17.5	0.5
Sperchios	647	1,662	89	779	13.7	0.41	7.91		+	3.1	2.3	30.3	18.4	45.3	0.6
V. Asopos	360	719	64	541	16.0	0.05	2.35		+	5.7	1.0	50.3	19.6	23.4	0.0
Zone 1 (rivers listed from west to east)															
Axios	752	24,398	417	603	10.1	5.92	7.70	+		1.8	6.7	41.5	2.1	47.4	0.5
Strymon	723	16,816	374	545	10.4	2.79	5.26	+		3.1	8.4	36.6	5.3	45.7	1.0
Nestos	1,043	6,213	273	621	8.5	1.68	8.57	+		1.6	6.3	15.8	4.3	71.0	1.0
Evros	409	53,025	559	622	11.2	8.18	4.89	+		3.9	3.6	55.9	0.9	34.7	1.0

E mean catchment elevation, *CA* catchment area, *L* river length, *P* mean annual precipitation, *T* mean annual air temperature, *Q* mean annual discharge, *SQ* specific discharge, *si* silicate, *carb* carbonate, *Ur* urban areas, *Gr* grassland, *Cr* cropland, *Fo* forest, *OW* open water

[a] CORINE 2006 (European Environmental Agency)

[b] http://www.worldclim.org/; Data for current conditions (~1950–2000); Hijmans et al. [1]

[c] European catchments and Rivers network system (Ecrins) Version 1 (European Environmental Agency)

[d] E-HYPE pan-European hydrological model (http://www.smhi.se/en/2.2139/e-hypeweb/)

[e] Intermittent in particular reaches

[f] Intermittent

similarly to other Mediterranean countries, is non-perennial as a result of dry climatic conditions and karstic geological background; a rough estimation showed that about 39% of the country's surface area is occupied by basins drained by non-perennially flowing water courses [7].

Regarding climatic, geological, hydrological and hydrochemical aspects, the Greek territory may be divided into three basic geographical zones: the north-eastern zone (zone 1), the north-central zone (zone 2) and the western zone (zone 3) [8, 9] (Fig. 1). Concerning climate, the three zones present distinct characteristics [10]; zone 3 is marked by maximum average precipitation, particularly in the north (more than 1,500 mm/a), contributing 1.5 times to the country's precipitation relative to its surface area (data: [2]), zone 1 presents minimum precipitation, whereas zone 2 presents intermediate precipitation and minimum air temperature [11]. Considering the whole territory, climatic conditions range from typical Mediterranean in the plains and towards the coast, to continental in mountainous areas, and show a strong N–S and E–W gradient with a southward, and a less significant eastward, increase in evapotranspiration [12]. Thus, in large parts of southern (Attika, Eastern Peloponnese and Crete) and eastern (Aegean Islands) Greece, semi-arid climatic conditions prevail [13].

Geotectonically, zone 3 belongs to the External Hellenides, which extend along the Ionian coast and are bound to the east by the Pindos Mt. range. They were dominated by the Alpine orogenesis and reveal a rather simple geotectonic structure made up of sedimentary sequences, predominately flysch and carbonates, imprinted by karstic features. Situated east of the Pindos Mt. range, zones 1 and 2 fit in the Internal Hellenides, which were additionally affected by older orogenetic movements, and reveal a complex geotectonic structure, dominated by metamorphic massifs, plutonic and volcanic intrusions and ophiolite suture zones. Geochemically, zone 1 belongs to an acid silicate type, zone 3 is termed a carbonate type zone, whereas zone 2 is characterised by a mixture of carbonate rocks and silicate rocks mainly of magmatic origin [8, 14].

Surface runoff presents high spatial variability due to the respective variability of climatic and geological features. Lotic systems are more abundant in zone 3 and are marked by high specific discharge (in general over 13 l/s*km^2) due to increased precipitation. However, specific discharge and the abundance of river networks diminish when going south, due to dryer climatic conditions (Table 1). As a result of carbonate substrates, river basins with minimum stream density predominate in this zone. In zones 1 and 2, surface runoff is scarcer. Rivers present low specific discharge (in general below 13 l/s*km^2) and many river stretches have temporary flow regimes especially towards the south and east. Due to the prevalence of impermeable silicate rocks in zone 1 basins, stream density is maximum compared to the other zones. Finally, the geologically mixed zone 2 hosts catchments with intermediate stream density [8, 9].

River basin geological, morphological and climatic characteristics shape morphological features of river and stream corridors; in zone 3, V-shaped valley forms with narrow floodplains and deep and wide riverbeds characterise the calcareous river basins. In zone 2, V-shaped valleys and riverbeds with medium width and depth predominate, while floodplains are rather narrow. Finally, zone 1 is dominated by water flows with U-shaped valley forms and wide floodplains but narrow and shallow riverbeds as a result of highly impermeable rock formations that dominate this zone [9].

Greek rivers tend to have naturally high sediment fluxes due to high relief ratios, high seasonal climatic variation, easily erodible rock formations, and sparse vegetation. Sediment fluxes have further increased by massive deforestation, fire, grazing and other human activities such as mining [8]. As a result of sediment transport and deposition, most major Greek rivers form deltaic plains. The Axios together with the adjacent Loudias and Gallikos Rivers form the most extensive deltaic area in Greece (600 km^2), followed by the Nestos Delta (434 km^2). The Arachthos River creates, together with Louros River, a large double delta of 350 km^2. The Evros Delta extends over 188 km^2 and the Alfeios Delta covers 113 km^2. The Sperchios River has a dynamically expanding delta of 196 km^2.

From a hydrochemical point of view, considering 39 river basins, ranging from small to very large, the three zones show differences in water temperature, conductivity, and calcium, magnesium, bicarbonate, sulphate and silicate concentrations [9, 15]; in general, zone 3 shows maximum water temperature due to its karstic nature, zone 1 is characterised by maximum sulphate concentration due to the dissolution of pyrite ores, followed by zone 3 which is affected by evaporite dissolution. Furthermore, zone 2 shows maximum silicate concentrations as a result of ophiolite weathering, followed by zone 1 rivers, which are affected by acid silicate weathering [8, 9].

Finally, from an ecoregional aspect, zone 1 corresponds to Ille's ecoregion 7, whereas zones 2 and 3 belong to ecoregion 6.

This review chapter illustrates abiotic water quality characteristics of Greek rivers and streams both in spatial and temporal scales, and attempts to interpret the origin and variability of their composition as it results from various natural factors and processes as well as human interference.

2 Materials and Methods

For the purposes of this chapter, the river database of the Institute of Marine Biological Resources & Inland Waters (IMBRIW) of the Hellenic Centre of Marine Research was utilised. The data originate from various European and national projects carried out by IMBRIW during the period 1996–2015 and refer to river and stream sampling stations dispersed throughout the country.

Regarding major ions (Ca^{2+}, Mg^{2+}, Na^+, K^+, HCO_3^-, SO_4^{2-}, Cl^-) and silicate,[2] data from 77 sampling stations (60 rivers and streams) were analysed, and the hydrochemical composition of mainland and island rivers and streams was

[2] For the specifications of laboratory analysis see [9].

compared. The majority of these data originate from IMBRIW's national and international projects, whereas 10% of the data set corresponds to long-term measurements of the Ministry of Agriculture for major rivers [16]. For physico-chemical characteristics (water temperature, conductivity, pH and dissolved oxygen), data from 394 stations (129 rivers and streams) were used. These data originate mainly from the National Monitoring Program (2012–2015) and other IMBRIW's national and international projects. Regarding nutrients (NO_3^-, NO_2^-, NH_4^+, TN, PO_4^3, TP)[2], data from 774 sampling stations (182 rivers and streams) were used, 97% of which originate from IMBRIW's national and international projects, including the National Monitoring Program (2012–2015). Lastly, 3% of these data were obtained from the Ministry of Agriculture [16] and other literature sources.

These data were statistically treated in order to gain an overview on the country's river hydrogeochemical and pollution profile.

3 Results & Discussion

3.1 Hydrochemical Composition and Classification

Due to the predominance of carbonate minerals in rocks, in recent sediments and in riverine particulate matter [17], the discharge-weighted concentrations of major ions in Greek rivers are far higher than both the world and European averages [3] due to elevated Ca-HCO_3 ion content. Mean conductivity is 457.7 μS/cm (Table 2) and mean Total Dissolved Ion (TDI) concentration is 374.9 mg/l (Table 3). Electrical conductivity ranges between 45 and 100 μS/cm for "pristine" headwater stream stations draining registrant rock types (mainly found in north-western mainland Greece and in Samothraki Island), and may reach > 900 μS/cm for polluted river sections, rivers that are fed by significant karstic inputs, or coastal stations affected by sea water intrusion. Greek rivers are in general alkaline in nature (mean pH: 8.03, Table 2) and only three out of 129 rivers are acidic (pH < 7).

Table 2 Basic statistics regarding water temperature, electrical conductivity and pH for 394 river stations distributed throughout the country (IMBRIW-database, period of measurements 2000–2015)

	T_w	EC	pH	DO
	°C	μS/cm		mg/l
Average	14.8	457.7	8.03	9.82
Median	14.9	392.7	8.04	10.03
Max	34.2	2740.1	9.47	14.91
Min	6.8	45.5	6.77	2.91
Stdev	3.1	309.4	0.41	1.63
CV	20.6	67.6	5.16	16.64

T_w water temperature, *EC* electrical conductivity, *DO* dissolved oxygen

Table 3 Basic statistics for major ions, silicate, TH and TDI (in mg/l) considering 76 river stations distributed throughout the country (IMBRIW-database, period of measurements 1980–2013)

	Ca^{2+}	Mg^{2+}	Na^{+}	K^{+}	HCO_3^{-}	CO_3^{2-}	SO_4^{2-}	Cl^{-}	SiO_2	TDI	TH
Average	55.33	17.61	14.12	2.55	201.05	7.85	47.24	17.43	11.75	374.94	224.54
Median	51.36	12.02	8.92	1.89	199.03	3.15	27.22	8.23	9.60	321.42	214.04
Max	158.89	69.15	94.04	10.14	473.12	44.26	332.79	136.58	100.75	1419.71	586.00
Min	4.15	1.17	0.65	0.78	19.70	0.00	3.90	1.19	0.10	31.64	150.00
StDev	32.77	17.75	16.90	2.01	96.00	11.01	61.23	24.59	8.50	270.76	132.82
CV	59.23	100.75	119.74	78.80	47.75	140.32	129.60	141.06	72.31	59.56	59.15
% Ion/TDI	14.76	4.70	3.77	0.68	53.62	2.09	12.60	4.65	3.13		

TH total hardness ($CaCO_3$), *TDI* total dissolved ions, *% Ions/TDI* percentage contribution of each ion to the TDI

The majority of Greek rivers belong to the calcium carbonate hydrochemical type ($Ca > Mg > Na > K–HCO_3 > SO_4 > Cl$, meq/l), similarly to the average of rivers worldwide [18], where bicarbonate is the dominant ion, followed by calcium [3]. However, there are certain rivers and streams with differing ion sequence [9]. For example, the Aoos headwaters which flows through mafic rocks are of a magnesium carbonate type ($Mg > Ca$), whereas, along the Evros, Acheloos, Pamisos, Kalamas, Florinis, and Onochonos, in the Nestos headwaters and tributaries (Arkoudoremma and Diavoloremma), and in Aliakmon headwaters, sodium is the second dominant ion, while chloride dominates in the Acheloos and Arachthos. Finally, in the Alpheios and Soulou rivers that drain lignite deposits, sulphate is the dominant anion.

As a result of geological and climatic controls, an increase in river mineralisation towards zone 3 is evident; on average, TDI concentration (excluding silicate) ranges from 221 mg/l in zone 1, to 297 mg/l in zone 2, and reaches 463 mg/l in zone 3 (Table 4). Thus, rivers located in zone 1 show a low mineralisation, rivers draining zone 2 basins are of a medium mineralisation, and those situated in zone 3 show high mineralisation. Total Hardness (TH) reveals a similar trend ranging, according to [28] classification, from hard (average, 140 mg/l $CaCO_3$) in zone 1, to very hard in zones 2 and 3 (210 and 269 mg/l $CaCO_3$, respectively) (for TDI and TH classification see [9]).

Rivers and streams adjacent to the sea (located both in the mainland and on islands) present differing composition. Situated within a semi-arid climatic zone, Aegean island streams are more mineralised that mainland ones as a result of higher air temperature and evaporation, and marine aerosol influence (Fig. 2).

3.2 *Factors Controlling River Water Composition*

A river is a four-dimensional (length*width*depth*time) open system with strong interactions between its drainage basin and the subsurface. Various abiotic and biotic processes, such as tectonic dynamics, weathering, erosion and sedimentation, evaporation, infiltration, flushing, and metabolic and biogeochemical processes, as well as human factors, interact within the watershed, the river floodplains, the riparian zone and the water body itself, determining its hydrochemical regime (e.g. [19]). These factors and processes are combined to create a diverse water composition that changes spatially and temporally.

Despite the dynamic interrelationships between factors and processes controlling river water composition, efforts are made to examine them separately in order to identify and weight the underlying driving forces that control aquatic quality composition and its variation in Greek rivers and streams.

Table 4 Basic statistics for water temperature, major ions, silicate, TH and TDI for 76 river stations situated in the three Greek geographical zones (IMBRIW-database, period of measurements 1980–2013)

	Ca^{2+}	Mg^{2+}	Na^{+}	K^{+}	HCO_3^{-}	CO_3^{2-}	SO_4^{2-}	Cl^{-}	SiO_2	TH	TDI
	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l
Zone 1 – 11 stations											
Average	34.84	7.61	14.08	3.78	119.20	6.94	36.58	10.14	11.41	139.48	244.58
Median	37.83	5.95	9.18	2.74	126.73	2.49	27.61	6.76	11.25	137.67	230.54
Stdev	20.78	5.32	10.98	2.71	60.09	10.51	31.84	9.80	1.88	97.38	153.91
CV	59.65	69.97	78.01	71.58	50.41	151.34	87.04	96.64	16.47	69.82	58.63
% Ion/TDI	14.25	3.11	5.76	1.55	48.74	2.84	14.95	4.15	4.66		
Zone 2 – 27 stations											
Average	39.38	18.75	6.79	2.34	189.59	8.42	23.82	7.57	14.14	209.89	310.81
Median	45.78	17.26	4.47	1.76	200.08	5.58	24.98	5.51	12.21	176.62	317.63
Stdev	18.93	18.03	5.67	1.93	98.00	10.62	17.85	6.28	9.17	136.73	186.47
CV	48.06	96.12	83.54	82.43	51.69	126.09	74.95	82.89	64.83	65.14	49.10
% Ion/TDI	12.67	6.03	2.18	0.75	61.00	2.71	7.66	2.44	4.55		
Zone 3 – 35 stations											
Average	76.59	19.65	19.63	2.23	238.83	7.81	70.72	27.71	9.54	268.96	472.71
Median	73.57	12.29	10.55	1.80	211.20	2.14	31.28	17.40	6.55	242.91	366.78
Stdev	33.02	19.29	21.68	1.57	82.25	11.59	80.33	32.00	9.14	123.61	290.87
CV	43.11	98.15	110.44	70.55	34.44	148.51	113.59	115.47	95.86	45.96	49.87
% Ion/TDI	16.20	4.16	4.15	0.47	50.52	1.65	14.96	5.86	2.02		

TH total hardness ($CaCO_3$), *TDI* total dissolved ions, *% Ions/TDI* percentage contribution of each ion to the TDI

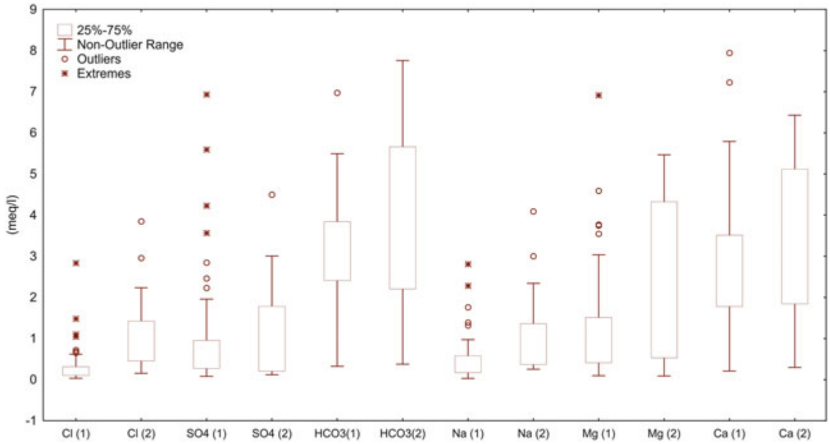


Fig. 2 Box plot diagrams for major ion concentrations in mainland (*1*) and island (*2*) rivers and streams for catchment areas up to 1,000 km^2

3.2.1 Atmospheric Inputs and River Water Quality

Atmospheric deposition is recognised as a potential source of major ions and nutrients to river ecosystems [20–22]. According to Table 5, which presents the ratios between ion concentrations in bulk precipitation to the ion concentrations of related river basins, the potential contribution of bulk precipitation to river solutes, and especially to nutrients, is remarkable. For major ions, the potential inputs of bulk precipitation are largely expressed in basins underlain by resistant rocks [23] drained by rivers with low solute concentrations and in headwater reaches with minor pollution pressures, which is the case in the Samothraki Island streams [24] and the Krathis River basin [25]. The high concentration of nutrients in bulk precipitation is affected by Greece's location in the eastern Mediterranean, since it receives air masses laden with anthropogenic nutrients from industrialised and agricultural areas of central and Eastern Europe [26].

Focusing on sodium and chloride, about 15% of their riverine content is provided by precipitation [15]. A significant portion of these ions, in rivers in the vicinity of the sea, is attributed to marine aerosol [25]. Thus, the ratio of chloride and sodium between island and mainland streams is at a maximum (3.7 and 2.6, respectively) compared to the respective ratios for other major ions (ranging between 1.9 for magnesium to 0.7 for potassium). Moreover, the Na/Cl ratio in island-streams (1.08) lies closer to the respective seawater ratio (0.86) than in the mainland ones (1.77).[3] The marine influence is also demonstrated by the

[3]It should be noted that mainland rivers are subject to anthropogenic pollution pressures more than island ones. Since municipal wastes enrich surface runoff predominately with chloride ions [15], the Na:Cl ratio in mainland rivers in absence of pollution would be even higher.

Table 5 Ratio between element concentrations in bulk precipitation and river water

	Ca	Mg	Na	K	HCO_3	SO_4	Cl	TDS	NO_3	NO_2	NH_4	TN	PO_4	TP	SiO_2	*n*
Acheloos R.	0.07	0.12	0.25	2.00	0.09	0.23	0.29	0.15	0.92	0.86	6.78	1.21	3.24	1.57		14
Krathis R.	0.16	0.09	0.93	0.78	0.15	0.14	–	0.16	2.6	5.23	24.44	5.73	27.34	5.26	0.10	13
Evrotas R.[a]	0.05	0.04	0.08	0.44	0.04	0.17	0.03	0.06	0.62	~1	40.85	1.38	>4.25	>5.27	<0.03	1
Samothraki streams[a]	0.55	0.62	0.38	1.38	0.83	0.31	0.40	0.58	0.29	>1.69	13.81	0.77	>3.75	>4.62	0.09	1

TDS total dissolved solids, *n* number of measurements
[a]Unpublished data

exponential decrease of these ions with altitude [9] that is consistent with the findings of Meybeck [27].

3.2.2 Geological Controls

On a global scale, the ultimate source of most dissolved ions is considered to be the mineral assemblage in rocks near the land surface [28]. Geology, in combination with microbial processes taking place in soils or in the water/sediment interface, controls the quality of solutes and affects their quantity together with discharge.

The three geographical zones of the country are differentiated by both climatic and geological features, therefore it is difficult to distinguish which is the dominant factor that determines the hydrochemical character of Greek rivers. Nonetheless, there are signs that spatial geological differences may overwhelm respective climatic variability in controlling solute concentrations [9].

Due to the predominance of carbonate minerals in the geological background of the country, the main factor controlling Greek river composition is carbonate dissolution, whereas other factors comprise acid and mafic rock weathering, pollution and biological activity, as it has been shown for major rivers [15]. Statistical analyses revealed that major ions, such as calcium, magnesium, hydrogen carbonate, sulphate, sodium and chloride, primarily correlate with the portion of recent (neogene and quaternary) sediments within river basins of major rivers [15]. In fact, considering 64 river basins (90 river stations), ranging from very small ($<10\ km^2$) to very large ($>10{,}000\ km^2$), a positive correlation between TDI and the portion of recent sediments in the river basins is evident (Fig. 3). This trend is initiated by the fact that recent sediments are more easily soluble than the respective bedrock itself. For the same reason, when going downstream, solute concentrations increase since recent sediments are more abundant at low altitudes [15]. However, as alluvial aquifers are commonly developed within these sediments, the downstream increase of solute concentrations is additionally triggered by the contribution of these alluvial aquifers (which are laden with excess solutes, and especially carbonate) to river runoff.

Moreover, rivers situated in the northern part of zone 3 (Fig. 1) reveal high mineralisation compared to rivers situated in the other zones. This occurs despite the fact that these river basins, characterised by low evaporation, maximum precipitation and maximum runoff coefficients (Table 1), demonstrate maximum dilution processes compared to the other zones. Also, the concentrations and the relative portion of major ions in the three zones (Table 4) mirrors the rock formations of river basins; silicate and magnesium, show maximum concentrations in basins with mafic rocks (zone 2) due to the weathering of olivine and serpentine minerals (e.g. [29]); the decrease of potassium concentration, when moving from zone 1 to zone 3, is attributed to the respective decrease of acid silicate rocks; sulphate shows higher concentration in particular river basins marked by evaporites (zone 3), lignite deposits (all zones), or with abundant pyrite ores (zone 1); rivers draining carbonate rocks reveal low phosphorus concentrations, since carbonate

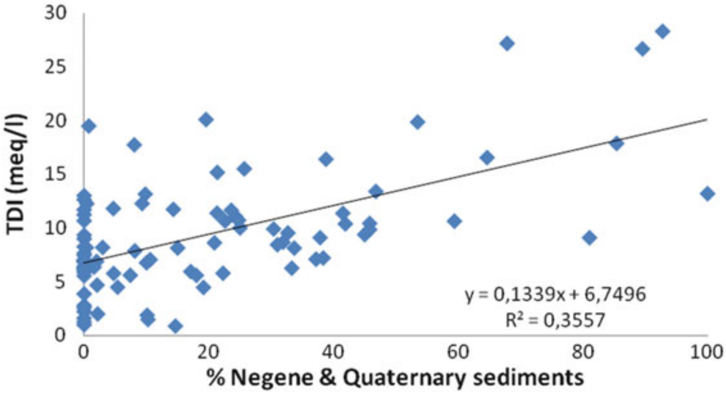


Fig. 3 Correlation between the portion of neogene and quaternary sediments within river basins and the respective TDI concentrations for 64 river basins (90 river stations) ranging from very small (<10 km^2) to very large (>10,000 km^2)

minerals may act as a phosphorus sink [8, 25]. Finally, the example of river Nestos, with vast differences in major ion composition between the magmatic upstream (low solute concentration with high alkali ion portion) and the calcareous mid stream reaches (high solute concentration with high earth alkali ion portion), illustrates the influence of geology on water quality within a single river basin [30].

Geological factors also affect river water composition through rock permeability. In fact, since surface/subsurface water interactions increase with increasing rock permeability, rivers draining alluvial formations are rich in solutes [15]. Particularly in summer, when river flow is practically base flow (in absence of precipitation), groundwater composition imprints river water [9].

Finally, geology also controls sediment transport; small mountainous basins with predominance of flysch, such as the Aoos and Arachthos, show high sediment transport rates that are comparable to large river basins [8].

3.2.3 Climatic Influence

Like other Mediterranean regions, Greece is marked by a distinct cool and wet season followed by a warm and dry period, while it is influenced by a sequence of regular and often extreme drying and flooding periods of unpredictable intensity and frequency [31].

By controlling air temperature, precipitation and evaporation rates, vegetation type and cover, as well as biogeochemical processes, spatial and temporal climatic variations affect the rivers' hydrological and hydrochemical regimes. Climate affects river runoff, which is a major driver for river water composition since it controls solute concentrations by dilution (Sect. 3.2.4). Considering a relatively homogenous geology within zone 3, climatic factors imprint river water composition; a southward increase in drought, dramatically diminishes river flow (Table 1), especially in summer, resulting to a solute concentration increase in Peloponnesian, and Aegean Island rivers and streams. Particularly in semi-arid areas, such as Attika and the Aegean Islands (zones 2 and 3), streams with low flow, or non-perennial hydrological regimes, dominate, and Island streams show significantly higher solute concentrations compared to more temperate continental basins (Fig. 2).

As a result of climatic influence, air and river water temperature increases with decreasing catchment latitude and altitude [9], thus enhancing the rates of biogeochemical and ecological processes (photosynthesis, respiration, mineralisation and decomposition) [32]. Climatic conditions and vegetation characteristics also affect soil properties; highland rivers or upstream river sections are more enriched with dissolved organic carbon than lowland river reaches [15, 33]. In addition, undisturbed forested catchments show increased TP during floods due to leaching of soils with high organic matter content [9]. Finally, as a result of climatic controls, the western part of the country (zone 3) is characterised by poorly leached soils [34].

Extended drought periods may dramatically affect river water composition; a drought wave that affected Europe at the end of the 1980s – beginning 1990s, in combination with excessive water withdrawals for irrigation, caused a dramatic reduction of river runoff in Greece, leading to a substantial rise of solute concentrations (Fig. 4). For example, the conductivity in Axios rose by ~40% compared with previous years [8, 35]. Droughts may also cause wildfires; in 2007, during prolonged summer heat waves, heavy wildfires destroyed extended forest and shrub land areas, causing an increase of surface runoff and erosion rates and elevated sediment and phosphorus concentrations in receiving water bodies [36]. These facts reveal the impact of droughts on river water quality, and demonstrate the potential impacts of future climate change on river hydrochemical regimes.

3.2.4 Hydromorphological Factors

Rivers' temporal hydrological regime is controlled by climatic (precipitation and evapotranspiration variations and trends) and geological factors (rock permeability). Discharge fluctuations control river water chemistry by determining the volume of water available to dilute concentrations (e.g. [28]), by influencing the amount of baseflow in river flow, and by initiating flood events and subsequent flushing processes (e.g. [37]).

The discharge regime of Greek rivers generally follows that of the rainfall and snow melting patterns, and consequently exhibits both strong seasonal and inter-annual variability. In this context, variable geographical distribution of climatic patterns, considerable geomorphological variations and complex geological and hydrogeological formations and structures that mark the country, give rise to a high spatial variety of river and stream hydrological regimes. Malikopoulos (in [38]) distinguishes five seasonal types of runoff regimes for major rivers; a simple rain type with one winter discharge peak, a complex rain type with two discharge peaks (where winter peak > autumn peak), a snow-rain type A (where winter

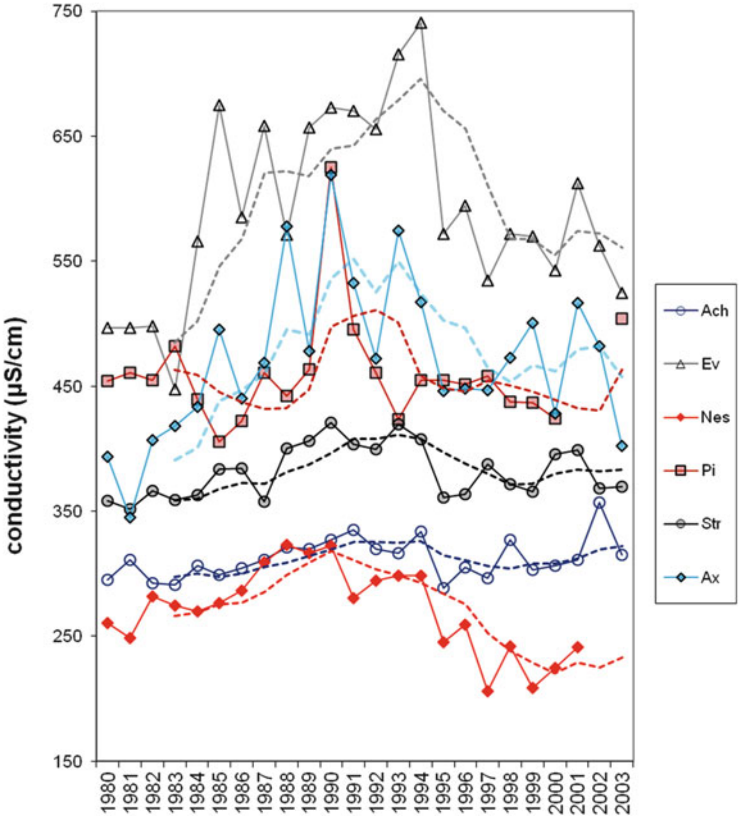


Fig. 4 Inter-annual variation of mean annual conductivity in major Greek rivers (data Ministry of Agriculture) and moving averages to smooth out short-term fluctuations and highlight longer-term trends or cycles. *Ach* Acheloos, *Ev* Evros, *Nes* Nestos, *Pi* Pinios, *Str* Strymon, *Ax* Axios, *Dotted lines* present 4-year period moving averages

peak > spring peak), a snow-rain type B (where spring peak > winter peak) and a spring type with one discharge peak.

All types, besides the spring type, representing karstic inputs that smooth seasonal variability, reveal a strong seasonal regime, mostly flushy in nature, and low summer flow. During summer (June–August), when rainfall is scarce, and autumn (September–October), when groundwater tables are sinking to minimum levels, surface runoff drops to a minimum and often river and stream stretches cease to flow, presenting a variety in hydroperiods ranging from intermittent to episodic. During the phase of desiccation, river reaches may be composed of a series of connected or isolated pools, or may dry out completely. This is a natural phenomenon under the dry climatic conditions and the karstic geology of large parts of the country [39]. However, temporary flow regimes are additionally triggered by water resource overexploitation as has been shown for the Evrotas River basin [40].

High baseflow and low rainfall contribution to river runoff smoothes seasonal hydrological variations and diminishes the occurrence of floods [8]. Thus, rivers imprinted by karstic features (where large amounts of water may be stored in subterranean aquifers) such as the Acheloos, the Louros, the Alfeios, the Evinos, the Pamisos, the Angitis, etc., show a low to moderate ratio between long-term monthly maximum and minimum discharge. The ratio in these rivers and the ones with regulated water courses, such as the downstream portions of Nestos, Aliakmon and Acheloos, ranges between 1.5 and 7. In contrast, the lower parts of Strymon and Evros, the upper Aliakmon, the Aoos, the Arachthos, the Pinios, the Sperchios, the Evrotas, etc., show high seasonal hydrological variations with ratios ranging between 8 and 20 and are prone to floods [8].

Hydrochemical variations depend on discharge fluctuations and the origin of waters (surface runoff, interflow, base flow) that contribute to the river hydrography. Thus, the seasonal regime of solute concentrations in river water is controlled by three main processes that are governed by hydrological factors: (1) dilution, during spring and selectively during winter, (2) concentration, due to evaporation and base flow contribution, during the dry season (in summer river water is practically represented by groundwater) and (3) enrichment due to flushing of soil-salts (e.g. [37, 41]), following flood events, occurring in autumn, winter and rarely in spring [15]. The relative importance of the aforementioned processes controls the intra-annual solute concentration variations in a river [14]. For example, the strong seasonal hydrological fluctuations in zone 1 rivers (due to the existence of small groundwater aquifers) cause respective seasonal solute variations [9].

As a result of dilution processes, specific conductance is lowest during spring (snowmelt) and winter (maximum rainfall) and peaks during base flow conditions. Thus, an inverse relationship between discharge and conductivity is commonly apparent (e.g. Fig. 5). Such rivers are of a “dilution type” [15]. In the case of the Nestos and Axios, the conductivity during low flow periods increases exponentially as a result of increased base flow contribution to river flow, intense evaporation, and pollution [3].

Impact of Droughts

During the dry period (June–September/October), river discharge is at a minimum, as a result of lack in precipitation and drop in ground water tables. Seasonal drought commonly leads to the formation of connected and disconnected pools or to complete desiccation of reaches even in the main stem of large rivers, especially during particularly dry years.[4] In the Evrotas River, where every year particular reaches dry out in summer, carbonate precipitation occurs due to increased

[4]When seasonal drought is artificial, i.e. is largely caused by water overexploitation, river ecosystems are seriously threatened leading to aquatic biodiversity losses [40].

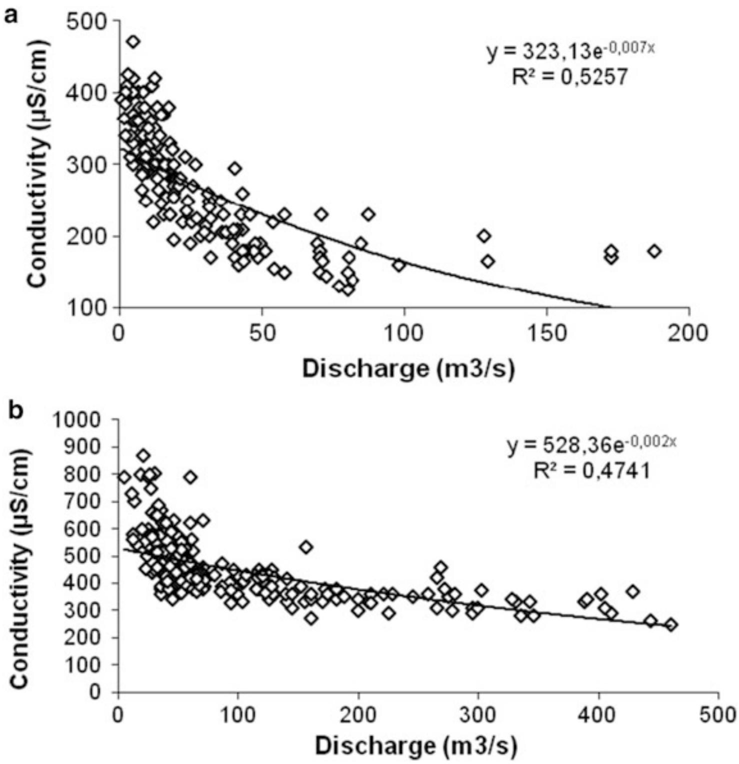


Fig. 5 Correlation between monthly instant discharge and instant conductivity in the free-flowing portion of the Nestos and Axios rivers according to long-term data series (Nestos: 1974–1993; Axios: 1977–1994) of the Ministry of Agriculture

photosynthesis, while denitrification and ammonification processes dominate in disconnected pools [39]. River flow (current velocity) controls the accumulation or release of particulate organic matter in marginal waters and in stream pools, thus influencing redox potential and mineralisation. These processes determine oxygen concentration and pH and affect the carbonate equilibrium as well as the concentrations and speciation of nutrients.[5] For example, in an intermittent river basin in Crete (Anapodaris River), in marginal waters, where current velocity was extremely low, sediment organic matter accumulated and dissolved oxygen approached zero, as a result of respiration [42].

[5]Most of these parameters may affect biota composition and abundance, thus impairing ecological quality of rivers (according to the WFD).

Impact of Floods

The concentration of autumn rainfall, in short but heavy storm events, creates flush floods. During these events, the extension of the drainage network to dry areas, where a store of readily soluble material is trapped, contributes to increasing concentrations of sediments, solutes and associated pollutants [43]. This mechanism enriches river water with sediments, major ions and pollutants. The impact of floods on aquatic quality may be detected even on a monthly basis [3, 25]. However, when using automatic gauging stations, the results of initial flush floods on river water composition are clearly visible; initial floods monitored in the Krathis and Evrotas rivers, revealed significant mobilisation of sediments and associated particulate nutrients, and specific nutrients surpassed qualitative standards [25, 39]. Regarding sediment transport, heavy initial autumn rains on desiccated soils often cause landslides, especially where unconsolidated sediments prevail. Thus, 50–95% of the annual sediment transport occurs during initial flood events [44].

Impact of Water Management

In the 1950s, the first large dams were constructed. Nowadays, 164 large dams (source: International Commission on Large Dams) regulate river runoff and most large and medium-large rivers are fragmented [8]. Most reservoirs have multipurpose functions (i.e. hydropower generation, irrigation, urban water supply, cooling thermoelectric plants, aquaculture, and recreation), with hydropower (H/P) generation and irrigation covering almost 70% and 30% of the usable volume of reservoirs, respectively. The most modified river is the Acheloos, in western Greece. In its headwater and middle sections, seven reservoirs exist (or are nearly completed), four of which are on the main stem. These reservoirs cover 150 km^2 with a total storage capacity of ~6.6 km^3. Moreover, ~0.15 km^3/year is transferred to the Pinios basin and an additional 0.6 km^3/year is planned to be transferred to the same basin. Four large reservoirs are located along the main stem of the Aliakmon River, three along the main channel of the Nestos River within the Greek territory, two along the Arachthos River, whereas other ten courses of major rivers are also fragmented [8]. From the major rivers, only the Sperchios and the Evrotas are free flowing.

Dam operation for H/P generation smoothes hydrological variations and modifies the hydrological regime, thereby contributing to flood mitigation and retention of sediments and pollutants. However, impounded rivers exhibit a disturbed hydrological regime downstream of the reservoirs; the Acheloos, Nestos and Aliakmon rivers exhibit high to maximum discharge in the summer period, due to the increased needs of H/P production for cooling purposes. Particularly for the Acheloos River, the second higher annual discharge peak occurs in July (which prior dam construction revealed the second lowest peak), whereas 30% of the annual runoff occurs during the summer months compared to 11% prior to dam construction [45]. This hydrological disturbance[6] initiates an abnormal hydrochemical regime downstream of the reservoirs; nowadays, in the Acheloos River, minimum solute concentrations are found in summer. This is due to both the lowering of carbonate concentrations as a result of photosynthetically driven carbonate precipitation occurring in the reservoirs, as well as to the outflow of deep, mineralised waters from the reservoirs during winter [45].

Agricultural land covers 30.2% of the Greek territory, of which 9.5% comprises irrigated farmlands, whereas agricultural water abstractions account for 89% of total abstractions [46]. Direct water abstractions take place from both lowland river reaches and headwater streams [7]. During dry years, farmers are affected by severe water shortages, particularly interruptions, during the irrigation season and are forced to construct provisional weirs on river courses for water abstraction. This, in combination with ground water over-pumping, favours artificial desiccation and many once perennial river stretches often cease to flow during the summer months [7, 40]. For example, in the Pinios River basin, intensive water use for agriculture (that exceeds 95% of the available water resources) deteriorated the water balance, making it strongly negative even in rainy years [47] and resulted in lowering of the groundwater tables by several tens of metres, thereby contributing to artificial desiccation of particular reaches during dry years (e.g. at the end of the 1980s beginning of the 1990s and in 2006/07). Also, the case of the Evrotas, with remarkable differences between the long-term rainfall reduction (10.5%) and discharge decrease (51.2%), is quite alarming, since it lost 84% of its initial discharge within three decades mainly as a result of water resources overexploitation [40]. Consequently, a vast proportion of its course becomes artificially non-perennial in particularly dry years; in October 2007, 80% of the river network dried out completely (Fig. 6), thus restricting irrigation ability and threatening endemic fish fauna [40].

The vast seasonal decline of river flow makes running waters particularly sensitive to anthropogenic pressures, especially regarding good water quality availability. In fact, many artificially non-perennial water courses suffer from eutrophication, hypoxia and high concentrations of industrial and agricultural contaminants [39, 48–50].

Regarding sediment transport, the long-term decline in river runoff, combined with an over 80% retention of sediments in reservoirs, resulted in a dramatic reduction of sediment fluxes during the past 50 years [8]. Consequently, deltaic areas of fragmented rivers are not expanding [44] or may have even started to decrease in size [51]. It is predicted that the sandy beaches and island barriers of the Acheloos Delta will gradually erode and coastal lagoons will be intruded by sea water [52]. Global sea level rise may further accelerate the destruction of many deltaic areas of the country.

[6] This abnormal hydrological behaviour downstream the reservoirs affects river bed, river bank and riparian areas' habitat characteristics, thus threatening biodiversity.

3.2.5 Effects of Physicogeographical Factors

Natural Controls

Altitude directly affects the equilibrium concentration of dissolved oxygen [53]. This may be the reason why nearly "pristine" upstream mountainous river reaches show oxygen undersaturation [54]. In addition, altitude indirectly affects hydrochemical properties of rivers; lowland rivers or river parts are marked by higher solute concentrations in comparison with upland reaches due to: (1) the higher base flow contribution on river flow, (2) the higher residence times of base flow in lowland aquifers, (3) the increase in floodplains and hence the enhancement of flushing processes, and (4) the higher concentration of population, mining, industrial and agricultural activities, creating pollution loads. These factors and processes, together with the downstream increase of recent sediment deposits, contribute to a respective enrichment of major ions and nutrients [8, 9, 15].

Slope controls current velocity, sediment transport and determines substrate and habitat composition. Small and medium sized river basins present an inverse relationship between water temperature and river valley slope [9]. This is attributed to hydrogeological factors, since rivers with steep slopes are characterised by fast running waters and runoff is almost exclusively represented by overland flow. Thus, steep river basins reflect more directly air temperature variations than those with smooth slopes of mixed (surface and subsurface) flow. Moreover, as a result of low base flow contribution in river flow, steep upland catchments include rivers with low solute concentrations [9].

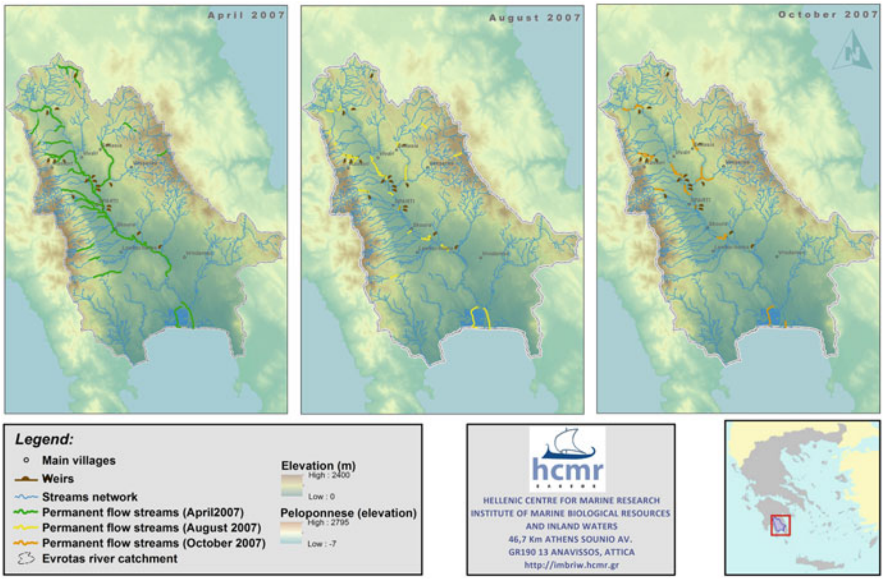


Fig. 6 Impact of the extreme drought of 2007 on the Evrotas River basin network

Anthropogenic Controls

Hydromorphological modifications (embanking, sand and gravel extraction, clear cutting of riparian vegetation in benefit to agriculture, straightening, reservoir building, etc.) also affect river water quality; the restriction of river floodplains and riparian vegetation limits natural attenuation causing river water pollution [55]. Moreover, the reduction of river flows and groundwater recharge in coastal areas leads to upstream sea water propagation and river water salinisation. On the contrary, healthy vegetation and soil cover act like a shield against pollution, as shown in the case of the Krathis River [25].

3.2.6 Effects of Pollution

Agricultural, industrial and municipal effluents cause an increase in nutrients, and organic and inorganic micropollutants in river water (see Sect. 4). Pollution also influences major ion composition; for example, rivers affected by lignite mining such as the Evros, Alpheios and Soulou show increased sulphate content, rivers affected by agriculture are enriched with potassium, whereas rivers influenced by municipal wastes show increased chloride and sodium concentrations [15]. Pollution also indirectly affects the carbonate content of river water; eutrophication and anoxic conditions produce excess CO_2 that may cause carbonate dissolution, thus enhancing the concentration of calcium, magnesium and bicarbonate in river water [56].

3.2.7 Biogeochemical Processes

Aquatic biogeochemical processes may be defined as the physical, chemical, geological and biological processes and reactions that govern the composition of the aquatic environment. In-stream processes include dissolution–precipitation, sorption–desorption, acid–base and redox reactions, and ecosystem metabolism. In addition, biogeochemical processes acting at the interface of water/sediment, surface/subsurface flow and within the riparian zone may also substantially shape the hydrochemical regime of rivers and streams in space and time.

Stream ecosystem metabolism is predominately apparent in natural and/or artificially eutrophic river reaches, such as shallow, slow-moving waters of large perennial rivers, in reservoirs and river deltas, in river pools during desiccation, and in river reaches affected by treated and untreated organic effluents (mainly municipal and agro-industrial wastewaters); for major Greek rivers, 12% of the variance of the data was explained by biological activity [15], showing that photosynthesis and respiration influence dissolved oxygen content and pH in eutrophic rivers. Additionally, small-medium river basins revealed a positive correlation between the excess of pCO_2 and the oxygen deficit [9] indicating the influence of biological activity on dissolved oxygen and carbon dioxide equilibrium in river water.

Metabolic processes, together with a series of biogeochemical processes, also influence carbonate equilibrium, and play an important role in nutrient and organic matter cycling. Photosynthesis causes oxygen and pH rise, carbonate oversaturation and precipitation [8, 15, 45, 54], whereas respiration initiates oxygen depletion, denitrification and ammonification[7] (e.g. [39, 42, 57, 58]).

Biogeochemical processes also control the natural attenuation of running waters; in the Krathis River "pristine" headwaters, almost no ionic N and P concentrations were found, despite the remarkable nutrient loadings of precipitation. This indicated the operation of various self-purification processes occurring in forest and riparian soils, at the soil/water interface or in-stream [25]. In contrast, in the almost "pristine" streams of Samothraki Island, nutrient levels may be affected by the reverse process; nutrient release from organic matter mineralisation, particularly leaves, accumulating within small pools that exist along river courses [24].

Wet-dry cycles control the activity of soil organisms, and thus, microbial biomass, mineralisation, denitrification, gaseous losses and ammonia volatilisation [59]. As runoff area increases, waters increasingly bypass biologically active soil profiles, and through leaching, mineralisation and nitrification processes, nutrients are flushed into the streams. Initial flood pulses that follow the dry period of the year create "hot moments" chiefly for suspended sediments and associated nutrients as well as for dissolved nutrients and major ions [25]. For example, flush peaks of 19 mg/l NH_4, that surpassed aquatic quality standards, were detected even in the unpolluted Krathis River during initial autumn floods as a result of rapid organic matter mineralisation and subsequent nitrification [25]. Finally, in the Evrotas River, initial flood events initiated a worsening of nitrate and nitrite quality [39].

4 The State of River Pollution

In Greece, mining and industrial activities are limited and locally restricted. The main anthropogenic pressures are connected with agriculture, i.e. agrochemical application, agro-industrial waste discharges and land clearing. Regarding municipal waste impact, over 90% of the country's population is connected to Waste Water Treatment Plants (WWTPs) (with 2/3 primary and 1/3 secondary treatments) [8]. However, small villages still maintain traditional sewage systems (permeable seep-tanks) and there is evidence of poorly functioning, or even not operating WWTPs in the smaller towns [8, 57].

According to the major rivers' long-term monitoring archives [16], river water shows a satisfactory oxygenation with minimum instant monthly values of dissolved oxygen ranging from 9.5 (Aliakmon) to 5.8 (Evros) mg/l. Dissolved oxygen (DO) concentrations <5 mg/l were only sporadically recorded

[7]Increased respiration in certain time periods may lead to oxygen depletion and ammonia production that threatens biota and may lead to, commonly reported, massive fish deaths.

Table 6 Basic statistics regarding nutrient concentrations (in mg/l) river sites (IMBRIW data base), the nutrient status classification, according to the Greek Nutrient Classification System (NCS, [9]), and the N/P ratios (by weight) for the Greek nutrient monitoring network

	$N\text{-}NO_3^-$	$N\text{-}NO_2^-$	$N\text{-}NH_4^+$	$P\text{-}PO_4^{3-}$	Total P	N/P
Nr of river stations	774	774	757	769	273	769
Mean	1.11	0.03	0.18	0.12	0.07	68.5
Median	0.58	0.01	0.03	0.03	0.03	18.4
Max	16.46	1.90	10.79	4.98	0.67	1979.8
Min	0.0015	0.0000	0.0007	0.0004	0.0019	0.07
Std dev	1.67	0.10	0.84	0.41	0.11	161.8
CV	150.2	367.3	463.1	334.9	159.9	236.3

N:P corresponds to: $(N\text{-}NO_3^- + N\text{-}NO_2^- + N\text{-}NH_4^+)/P\text{-}PO_4$(by weight)

(e.g. Evros: 4.2% of all measurements) [8]. According to the IMBRIW's data (394 stations – 129 rivers and streams), the mean DO concentration is high (9.82 mg/l) (Table 2) and only 7 stations show a DO value below 5 mg/l. However, as the case of the Sparta WWTP revealed, downstream of inadequately operating WWTPs, oxygen may temporarily drop to zero values [57].

Regarding nutrient pollution, according to the data of the Ministry of Agriculture [16], nitrate reached maximum long-term average concentration in the Evros (3.5 mg/l $N\text{-}NO_3$) followed by the Pinios and Axios (1.92 and 1.86 mg/l $N\text{-}NO_3$, respectively). Mean ammonia levels ranged between <36 (Aoos) and 140 (Aliakmon) μg/l $N\text{-}NH_4$, and TP was generally below 80 μg/l, and only the Evros and Axios revealed excessive levels (668 and 634 μg/l, respectively) [8]. Table 6 presents basic statistics regarding the nutrient levels (and the mean and median nutrient status classification) from the IMPRIW data base, including 774 river sites. Figure 7 illustrates the quality classification of the monitoring stations, based on nutrient concentrations, according to the Greek Nutrient Classification System (NCS) [9]. According to Fig. 7, 58% of the stations reveal high and good nutrient quality, whereas 42% of the stations show a nutrient quality that is below the good quality standards. In addition, nitrate, nitrite, ammonia and phosphate, in 29%, 34.8%, 45.5% and 71.7% of the examined stations, respectively, present reference conditions.

In general, nitrogen reference conditions in Greek rivers are low, as nutrient classification systems developed for the WFD purposes indicate [9, 42]. This is attributed to the mountainous relief with limited pollution pressures at medium and high elevations, to the poor soil organic matter of the country, and to natural attenuation processes [8, 25]. On the contrary, phosphorous reference conditions are relatively high compared to other European countries, possibly due to high geochemical background levels.

The range of nitrate concentrations in Greek rivers (0.0015–16.46 mg/l $N\text{-}NO_3$) resembles the range of nitrate concentrations found in European streams and rivers, which vary between <0.002 in least disturbed catchments, to over 14 mg/l $N\text{-}NO_3$

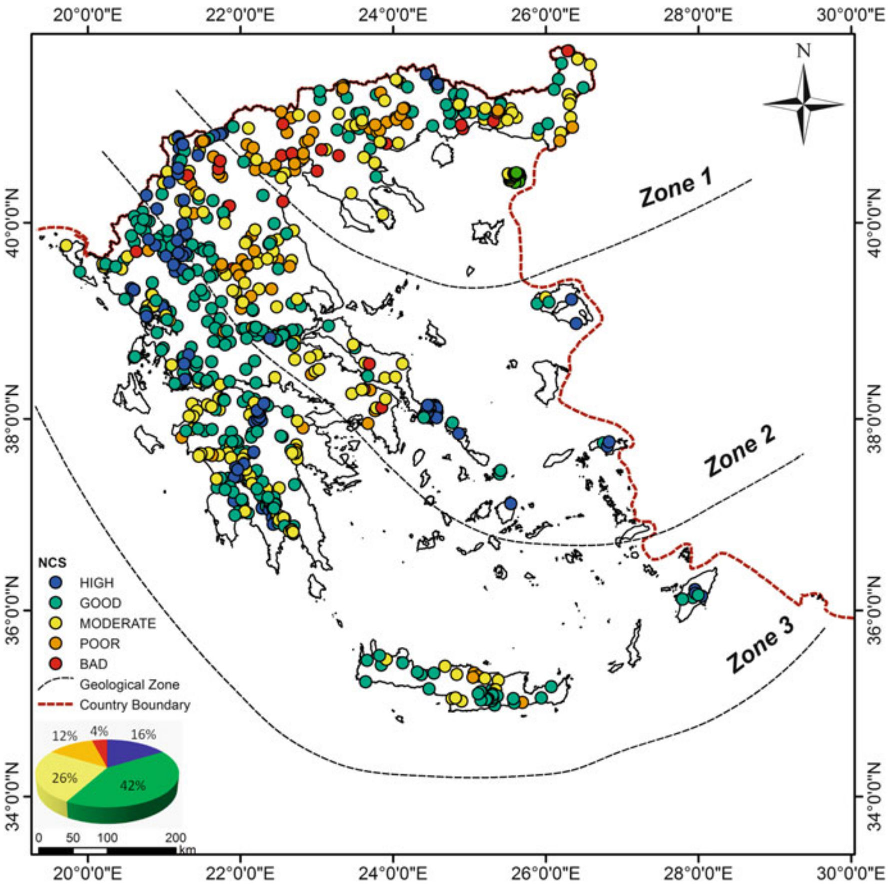


Fig. 7 River status characterisation according to the Nutrient Classification System (NCS) for Greek rivers [9] (data: IMPRIW data base, 774 river stations)

in the most intensively farmed catchments [60]. However, the mean nitrate concentration of Greek rivers (1.11 mg/l N-NO_3) is half of the mean nitrate concentration (2.2 mg/l N-NO_3) found in European rivers ([61], data 2010). Regarding phosphate, the mean concentration of Greek rivers (120 μg/l) exceeds the European average of 78.8 μg/l (1,379 rivers from 33 countries; http://www.eea.europa.eu/soer-2015/countries-comparison/freshwater, accessed March 2016).

According to Fig. 7, in zone 3, and in the western part of zone 2, good and high nutrient quality prevails, whereas in the eastern part of zone 2 and in zone 1, rivers with moderate and poor nutrient quality predominate. Besides the higher discharge that characterises rivers and streams draining western Greece, thus contributing to a dilution of pollutants, the predominance of carbonate geology in western Greek basins may cause retention of phosphorous in the solid phase [25]. On the contrary, rivers draining central Greece, such as the Voiotikos Asopos and the Pinios, show maximum pollution, whereas interregional rivers entering the country, such as the Evros and the Axios are among the most polluted rivers in the Balkans [8].

According to the prescriptions of OECD [62], average and median N/P ratios, as well N/P ratios in the majority of the monitoring stations (61%) present phosphorous-limited photosynthesis (TNin/P-PO4 > 12, by weight). 19% of the stations show a nitrogen-limited photosynthesis (TNin/P-PO4 < 5, by weight), whereas in 20% of the stations N/P ratios lie between 5 and 12. This finding agrees with the general consensus that, in freshwater systems, phosphorus is the primary limiting nutrient, with limitation by nitrogen playing a secondary role [63].

Overall, nutrients exhibit a downstream increase in their concentrations caused by a cumulative increase in human pressures, with some exceptions due to the contribution of local point pollution sources. For example, the Axios used to be significantly polluted with ammonia and phosphorus at the cities Veles and Skopje, as a result of municipal and industrial inputs, whereas further downstream water quality improved [8].

Despite the high geochemical background, especially in river basins situated in zones 1 and 2, dissolved heavy metal levels are generally low, compared to world averages and background levels [8]. Elevated concentrations occur in a number of cases due to a variety of factors, including intense agricultural applications, meteorological events, industrial effluents, mining activity, and the geochemical background [64]. Of the Greek rivers, the Axios is one of the most polluted rivers due to mining and industrial sources in FYR Macedonia[8] and the geochemical environment. Other rivers severely threatened by metal pollution are the Voiotikos Asopos and the Chalkidiki streams, affected by industrial and mining activities, respectively.

Regarding pesticide residues, based on the review of Lambropoulou et al. (this issue), who considered 19 rivers, the following results may be outlined: (1) river contamination by pesticides follows similar concentration levels and patterns as reported in most European countries, (2) the levels of some compounds decreased with time (e.g. organochlorines, atrazine, alachloretc), mainly due to a ban or the implementation of good agricultural practices; (3) in some areas with intense agricultural practices the concentrations of pesticides were in non-compliance with the environmental quality standards (EQS, Directive 2008/105/EC); (4) the ecological risk ranged from negligible to high depending on the pesticide and the target organism. For example, regarding algae, herbicides showed low to high risk, while insecticides showed negligible risk. Herbicides showed negligible risk for invertebrates (Daphnia magna) and fish (rainbow trout), whereas insecticides presented low to high risk for invertebrates.

[8]According to recent information, polluting industries in FYROM have been closed down.

5 Conclusions

River hydrogeochemical properties predominately derive from a combination of river basin geological (weathering, erosion, and rock permeability) and climatic factors (precipitation and evapotranspiration). Due to the distinct longitudinal geological and climatic characteristics of the country, Greek rivers are grouped within three N–S extending zones (i.e., from east to west, zone 1, zone 2 and zone 3) with certain enrichment in specific ions and increasing mineralisation towards the west. In zone 3, a southward increase in aridity causes a respective rise in non-perennial river courses and an increase in solute concentrations, especially in the semi-arid central and southern Aegean islands.

Geological factors, such as the existence of neogene and quaternary sediments in river basins, essentially control solute concentrations. Therefore, river basins with high recent sediment portion as well as downstream river sections reveal high solute concentrations due to the high solubility of these sediments, the contribution of groundwater aquifers (which are commonly developed within these sediments) to surface runoff, and the impact of pollution (since human activities are mainly developed on the lowlands which are commonly covered by sediments). On the other hand, climatic factors such as multi-year drought periods cause the deterioration of surface runoff and a substantial rise in river water salinisation. In addition, bulk precipitation potentially contributes significant concentrations of potassium, nitrogen and phosphorus to river water.

Geological, climatic and morphological factors control river runoff and the rivers' hydrological regime. The volume of water available, and thus discharge fluctuations, determine solute concentrations. Hydrological variations, coupled with hydrogeological factors, control surface/subsurface water interactions; during summer, river water is practically represented by base flow, which imprints river water composition, whereas flood events, especially in autumn, enrich river water with sediments and associated salts and pollutants in particulate and dissolved forms. Morphological factors alone determine human activities, surface/subsurface interactions and flood events, thus indirectly, affecting hydrological, hydrogeological and hydrochemical properties of rivers.

Land use and water management infrastructures and practices, combined with climate variability and change, caused a dramatic long-term river flow diminishing and there is evidence of artificial desiccation of river reaches during the summer period. Drying out processes affect biogeochemical and metabolic processes, and threaten water quality and ecological attributes. Adverse ecological consequences also result from dam operation for H/P production, which may reverse the downstream hydrological and hydrochemical regime.

Metabolic and biogeochemical processes affect river water composition, predominately in slow-flowing river reaches, marginal standing waters, and in reaches with connected or disconnected pools during summer. The occurrence of organic matter due to pollution (e.g. due to malfunctioning WWTPs and/or agro-industrial effluents) enhances those processes. Intense photosynthesis and respiration control dissolved oxygen and carbon dioxide, carbonate dissolution and precipitation, nitrification, denitrification and ammonification. Biogeochemical processes also contribute to the natural attenuation of river waters.

Regarding pollution, Greek rivers present lower reference levels for N-species compared to other European countries, and half the nitrate concentration compared to the European mean. On the contrary, phosphorus shows both higher mean and higher reference condition concentrations compared to other European rivers, possibly due to the higher geochemical background of the country. Despite the high geochemical background, heavy metal concentrations are generally low, compared to the world's averages, besides a number of river basins subject to intense mining and industrial activities (e.g. the Evros, Axios, Voiotikos Asopos and the Chalkidiki streams). Finally, pesticides follow similar concentrations and patterns as in most other European rivers, and in some rivers exceed environmental quality standards.

References

1. Hijmans RJ, Cameron SE, Parra JL, Jones PG, Jarvis A (2005) Very high resolution interpolated climate surfaces for global land areas. Int J Climatol 25:1965–1978
2. Ministry of Development (2003) Master plan for the management of Greek water resources. Directorate of Aquatic and Natural Resources, Athens, 519 pp (in Greek)
3. Skoulikidis N, Bertahas I, Koussouris T (1998) The environmental state of freshwater resources in Greece (rivers and lakes). Environ Geol 36(1–2):1–17
4. UNEP/MAP (2003) Riverine transport of water, sediments and pollutants to the Mediterranean Sea. UNEP/Mediterranean Action Plan, Athens, Greece, 111 pp
5. Ludwig W, Dumont E, Meybeck M, Heussner S (2009) River discharges of water and nutrients to the Mediterranean and Black Sea: major drivers for ecosystem changes during past and future decades? Prog Oceanogr 80:199–217
6. Vörösmarty CJ, Green P, Salisbury J, Lammers RB (2000) Global water resources: vulnerability from climate change and population growth. Science 289:284–288
7. Skoulikidis N, Sabater S, Datry T, Morais M, Buffagni A, Dörflinger G, Zogaris S, Sánchez-Montoya M, Bonada N, Kalogianni E, Rosado J, Vardakas L, De Girolamo A-M, Tockner K (2017) Non-perennial Mediterranean rivers in Europe: status, pressures, and challenges for research and management. Sci Total Environ 577:1–18
8. Skoulikidis N (2009) The environmental state of rivers in the Balkans - a review within the DPSIR framework. Sci Total Environ 407:2501–2516
9. Skoulikidis N, Amaxidis Y, Bertahas I, Laschou S, Gritzalis K (2006) Analysis of factors driving stream water composition and synthesis of management tools – a case study on small/medium Greek catchments. Sci Total Environ 362:205–241
10. Katsiou K, Kontaxopoulos G, Petrou MF, Trezos CG (1989) Snow loading in Greece--a statistical analysis. Techn Chron 9(1):37–70 (in Greek)
11. Mavromatis GN (1980) The bioclima of Greece. Relationships between climate and vegetation—bioclimatic maps. Athens Institute for Forestry Research, 63 pp (in Greek)
12. Dalezios NR, Loukas A, Bampzelis D (2002) Spatial variability of reference evapotranspiration in Greece. Phys Chem Earth 27:1031–1038
13. Yassoglou NJ, Catacousinos D, Kouskolekas A (1962) Land use in the semi-arid zone of Greece. In: Land use in semi-arid Mediterranean climates, UNESCO, International Geographical Union Symposium, Iraklion (Greece), 19–26 September 1962, pp 63–67

14. Skoulikidis N (2000a) Characteristics and types of Greek river basins. In: Proceedings of the 6th Pan-Hellenic Symposium of Oceanography and Fisheries, 23–26 May 2000, Chios, Greece, pp 206–211 (in Greek with English abstract)
15. Skoulikidis N (1993) Significance evaluation of factors controlling river water composition. Environ Geol 22:178–185
16. Ministry of Agriculture (2001) Qualitative characteristics of the country's rivers and lakes. Ministry of Agriculture, General Directory of Land Reclamation Works and Agricultural Structures, Athens, January 2001
17. Skoulikidis N (1990) Biogeochemistry of major Greek rivers. PhD Thesis, University of Hamburg, 313 pp (in German)
18. Meybeck M (1981) Pathways of major elements from land to ocean through rivers. In: Martin JM, Burton JD, Eisma D (eds) River inputs to ocean systems. Proc Worksh UN, UNEP, IOC, SCOR, Rome, pp 18–30
19. Webb BW, Walling DE (1996) Water quality. II. Chemical characteristics. In: Petts G, Calow P (eds) River flow and channel forms. Blackwell Science, Oxford, pp 102–129
20. Neal C, Skeffington R, Neal M, Wyatt R, Wickham H, Hill L, Hewitt N (2004) Rainfall and runoff of the Pang and Lambourn, tributaries of the River Thames, south-east England. Hydrol Earth Syst 8(4):601–613
21. Neal C, Reynoldsb B, Neal M, Hill L, Wickhama H, Pugh B (2003) Nitrogen in rainfall, cloud water, throughfall, stemflow, stream water and groundwater for the Plynlimon catchments of mid-Wales. Sci Total Environ 314–316(2003):121–151. doi:10.1016/S0048-9697(03)00100-1
22. Neal C, Reynoldsb B, Neal M, Hughes S, Wickhama L, Hill P, Rowland BP (2003) Soluble reactive phosphorus levels in rainfall, cloud water, throughfall, stemflow, soil waters, stream waters and groundwaters for the Upper River Severn area, Plynlimon, mid Wales. Sci Total Environ 314–316(2003):99–120. doi:10.1016/S0048-9697(03)00099-8
23. Bluth GJS, Kump LR (1994) Lithologic and climatic control of river chemistry. Geochim Cosmochim Acta 58:2341–2359
24. Skoulikidis N, Lampou A, Karaouzas I, Gritzalis K, Lazaridou M, Zogaris S (2014) Stream ecological assessment on an Aegean island: insights from an exploratory application on Samothraki (Greece). Fresen Environ Bull 23(5):1173–1182
25. Skoulikidis N, Amaxidis Y (2009) Origin and dynamics of dissolved and particulate nutrients in a minimally disturbed Mediterranean river with intermittent flow. J Hydrol 37:218–229
26. Kouvarakis G, Mihalopoulos N, Tselepides A, Stavrakakis S (2001) On the importance of atmospheric inputs of inorganic nitrogen species on the productivity of the eastern Mediterranean Sea. Global Biogeochem Cycles 15(4):805–817
27. Meybeck M (1996) River water quality. Global ranges, time and space variabilities, proposal for some redefinitions. Verh Internat Verein Limnol 26:81–96
28. Hem JD (1985) Study and interpretation of the chemical characteristics of natural waters. US Geological survey Water Supply Paper 2254, 3rd edn, 255 pp
29. Drever JI (1982) The geochemistry of natural waters. Prentice-Hall, Inc., Englewood Cliffs
30. Skoulikidis N (1991) Parameters influencing the chemical composition and pollution of river water – characteristic example the River Nestos. In: 2nd international conference on environmental science and technology, Molivos, Lesvos, 2–5 September 1991, pp 600–611
31. Gasith A, Resh VH (1999) Streams in Mediterranean climate regions: abiotic influences and biotic responses to predictable seasonal events. Annu Rev Ecol Syst 31:51–58. doi:10.1146/annurev.ecolsys.30.1.51
32. Ducharne A (2007) Importance of stream temperature to climate change impact on water quality. Hydrol Earth Syst Sci Discuss 4:2425–2460
33. Schlessinger WH, Melack JM (1981) Transport of organic carbon in the World's Rivers. Tellus 33:172–187
34. Nakos G (1984) Relationships of bio-climatic zones and lithology with various characteristics of forest soil in Greece. Plant Soil 79:101–121

35. Skoulikidis N (2000b) Impact of desertification processes on Greek river systems. In: International Conference on Mediterranean Desertification, 29 October–1 November 2000, Crete, Greece, vol 2, pp 277–285
36. Blake WH, Theocharopoulos SP, Skoulikidis N, Clark P, Tountas P, Hartley R, Amaxidis Y (2010) Wildfire impacts on hillslope sediment and phosphorus yields. J Soil Sediment 10:671–682
37. Walling DE, Foster IDL (1975) Variations in the natural chemical concentration of river water during flood flows and the lag effect: some further comments. J Hydrol 26:237–244
38. Therianos A (1974) The geographical distribution of the Greek rivers supply. Bull Greek Geol Soc 11:28–58 (in Greek with English abstract)
39. Skoulikidis N, Vardakas L, Amaxidis Y, Michalopoulos P (2016) Processes controlling aquatic quality during drying and rewetting events in a Mediterranean temporary river. Sci Total Environ 575:378–389
40. Skoulikidis N, Vardakas L, Karaouzas I, Economou A, Dimitriou E, Zogaris S (2011) Water stress in Mediterranean lotic systems; impacts and management implications in an artificially intermittent river (Evrotas River, Greece). Aquatic Science Special issue: "Recent Perspectives on Temporary River Ecology" 73:581–597. doi:10.1007/s00027-011-0228-1
41. Edwards AC (1973) The variation of dissolved constituents with discharge in some Norfolk rivers. J Hydrol 18:219–242
42. Skoulikidis N (2008) Defining chemical status of a temporal Mediterranean River. J Environ Monit 10(7):842–852
43. McClain ME, Boyer EW, Dent CL, Gergel SE, Grimm NB, Groffman PM, Hart SC, Harvey JW, Johnston CA, Mayorga E, McDowell WH, Pinay G (2003) Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems. Ecosystems 6:301–312
44. Poulos S, Collins M, Evans G (1996) Water-sediment fluxes of Greek rivers, southeastern Alpine Europe: annual yields, seasonal variability, delta formation and human impact. Z Geomorphol 40(2):243–261
45. Skoulikidis N (2002) Hydrochemical character and spatiotemporal variations in a heavily modified river of Western Greece. Environ Geol 43(7):814–824
46. Eurostat (2003) Farm Structure Survey 2003. Statistical Office of the European Communities Eurostat, Luxemburg
47. Loukas A, Mylopoulos N, Vasiliades L (2007) A modelling system for the evaluation of water resources management strategies in Thessaly, Greece. Water Resour Manag 10:1673–1702
48. Cooper SD, Lake PS, Sabater S, Melack JM, Sabo JL (2013) The effects of land use changes on streams and rivers in Mediterranean climates. Hydrobiologia. doi:10.1007/s10750-012-1333-4
49. Lopez-Doval JC, Ginebreda A, Caquet T, Dahm CN, Petrovic M, Barcelo D, Munoz I (2013) Pollution in Mediterranean-climate rivers. Hydrobiologia 719:427–450
50. Skoulikidis N (1999) The importance of factors controlling (extreme) qualitative variations of the country's surface waters in the frame of integrated river basin management. In: Proceedings of the 4th National Conference of the Hellenic Committee for the Management of Water Resources (EEDYP) "Management of Water Resources in Sensitive Areas of the Greek Territory", Volos, 17–19 June 1999, B': 13–17
51. Stournaras G (1998) Assessment of morphometric parameters of Greek Rhone-type deltas. Hydrogeologic and environmental aspects. Environ Geol 38(1):53–58
52. Bouzos D, Zelelidis A, Kontopoulos N (1994) Holocene and Plio-Pleistocene depositional environments of deltaic platforms. Examples from Western Greece. Bull Greek Geol Soc XX:385–396 (in Greek with English abstract)
53. Knoben RAE, Roos C, Van Oirschot MCM (1995) Task force on monitoring & assessment. Biological Assessment Methods for Watercourses, UN/ECE, 3, 86
54. Skoulikidis N (2002) Typological and qualitative characteristics of Greek-interregional Rivers. Mediterr Mar Sci 3(1):79–88
55. Nikolaidis N, Skoulikidis N, Papadoulakis V, Tsakiris K, Kalogerakis N (2009) Management plans for the agricultural basin of Evrotas River, Technical Report. In: Nikolaidis N,

Kalogerakis N, Skoulikidis N (eds) Environmental friendly technologies for rural development. LIFE ENVIRONMENT, LIFE05 ENV/GR/000245 (EnviFriendly)

56. Cicerone DS, Stewart AJ, Roh Y (1999) Diel cycles in calcite production and dissolution in an eutrophic basin. Environ Toxicol Chem 18:2169–2177
57. Lampou A, Skoulikidis N, Papadoulakis V, Vardakas L (2015) Evaluation of the waters' condition in the final receiver of the wastewater treatment plant in Municipality of Sparta. In: Proceedings of the 11th Panhellenic Symposium of Oceanography & Fisheries, Mytilini 13–17 May 2015, pp 661–664
58. Skoulikidis N, Gritzalis K, Kouvarda T (2002) Hydrochemical and ecological quality assessment of a Mediterranean river system. GlobalNest 4(1):29–40
59. Austin AT, Yahdjian L, Stark JM, Belnap J, Porporato A, Norton U, Ravetta DA, Schoeffer SM (2004) Water pulses and biogeochemical cycles in arid and semiarid ecosystems. Oecologia 141:221–235
60. Durand P, Breuer L, Johnes PJ, Billen G, Butturini A, Pinay G, van Grinsven H, Garnier J, Rivett M, Reay DS, Curtis C, Siemens J, Maberly S, Kaste O, Humborg C, Loeb R, de Klein J, Hejzlar J, Skoulikidis N, Kortelainen P, Lepisto A, Wright R (eds) (2011) Nitrogen processes in aquatic ecosystems. Cambridge University Press, Cambridge, pp 126–146. ISBN 9781107006126
61. EEA (2012) European waters – assessment of status and pressures. EEA Report No 8/2012. European Environmental Agency, Copenhagen
62. OECD (1982) Eutrophication of waters: monitoring, assessment and control. Organisation for Economic Cooperation and Development, Paris
63. Dolman AM, Wiedner C (2015) Predicting phytoplankton biomass and estimating critical N:P ratios with piecewise models that conform to Liebig's law of the minimum. Freshw Biol 60:686–697
64. Lekkas T, Kolokythas G, Nikolaou A, Kostopoulou M, Kotrikla A, Gatidou G, Thomaidis NS, Golfinopoulos S, Makri C, Babos D, Vagi M, Stasinakis A, Petsas A, Lekkas DF (2004) Evaluation of the pollution of the surface waters of Greece from the priority compounds of List II, 76/464/EEC Directive, and other toxic compounds. Environ Int 30:995–1007

Long-Term Hydrologic Trends in the Main Greek Rivers: A Statistical Approach

Angeliki Mentzafou, Elias Dimitriou, and Anastasios Papadopoulos

Abstract The scope of this research effort was to examine the effect of water management practices and land use changes on river flow over the last 3 decades, to identify the dominant trends in the discharge and precipitation time series and to examine the interrelationship between these two parameters. In order to accomplish these aims, the annual discharge time series of seven (7) major rivers in Greece were compared to the annual precipitation of the corresponding watersheds. This comparison was achieved through trend analysis of each time series, which involves the determination of basic statistical characteristics (normality, homogeneity, stationarity). Due to lack of satisfactory discharge time series at the downstream parts of each catchment examined, the results from E-HYPE pan-European hydrological model was used (European – HYdrological Predictions for the Environment). The main outcome of this work concludes that there is no consistent, single trend for the entire study period for any of the investigated rivers, while there are some wet and dry periods in the data which are very clear in all catchments and coincide at a temporal level. The main dry periods were at the end of the 1980s and the beginning of the 2000s. There is also a prolonged wet period during the last decade for all study catchments.

Keywords E-HYPE hydrological model, Management practices, Precipitation, River discharge, Trend analysis

---

A. Mentzafou (✉), E. Dimitriou, and A. Papadopoulos
Institute of Marine Biological Resources and Inland Waters, Hellenic Centre for Marine Research, 46.7 km Athens-Sounio Av., 19013 Anavissos, Greece
e-mail: angment@hcmr.gr

N. Skoulikidis et al. (eds.), *The Rivers of Greece: Evolution, Current Status and Perspectives*, Hdb Env Chem (2018) 59: 129–166, DOI 10.1007/698_2015_446,

Contents

1 Introduction

The need for sustainable management of water resources has become urgent in the recent decades, due to the substantial decrease of good-quality freshwater resources, mainly due to over-exploitation and the increasing pollution rates [1]. However, the usual water management practices at local and wider regional level are incomplete and are intended primarily to meet the sectorial needs that bring direct economic benefits, but also include environmental costs and possible long-term economic consequences [2]. In Greece, a Mediterranean country with relatively reduced rainfall and therefore moderate to low water resources [3], with a considerable spatial and temporal heterogeneity [3, 4], the often poor water management practices have resulted in water scarcity and desertification [5].

Climate change is also another potential adverse factor that can impose difficulties in developing and applying efficient water management practices that can mitigate the impacts from climate change themselves and ensure economic development and environmental preservation [6]. The Water Framework Directive (2000/60/EC) is a very important legislative framework that attempts to establish new scientific approaches and practices for designing and applying water management plans with its primary target the achievement of a good ecological status in the European waterbodies by the year 2015. Nevertheless, this aim in Greece has been significantly delayed due to socio-economic, cultural, political and scientific problems that obstruct the development of the appropriate tools and methods for the establishment of acceptable water management plans (e.g. [7]).

One of the important difficulties for developing and implementing water management plans is the low level of understanding of the aquatic systems functioning and the lack of appropriate quantification of the relevant processes. To overcome this problem, challenging scientific work has to be undertaken, requiring considerable human resources, sufficient field measurements and long-term, accurate environmental data. Moreover, the lack of appropriate data and the high cost of acquisition have led many scientists to attempt simulation of physical processes and their parameterisation based on statistical methods and in combination with other traditional scientific techniques [8].

The scope of this research effort was to examine the effect of water management practices and land use changes on river flow over the last 3 decades, to identify the dominant trends in the discharge and precipitation time series and to examine the interrelationship between these two parameters. In order to accomplish these, the annual discharge time series of seven (7) major rivers in Greece were compared to the annual precipitation of the corresponding watersheds. This comparison was achieved through trend analysis of each time series, which involves the determination of basic statistical characteristics (normality, homogeneity, stationarity), in order to select the appropriate data analysis procedure to the given time series and to avoid incorrect conclusions. In order to determine the basic statistical characteristics (normality, homogeneity, stationarity and trend) of annual discharge and precipitation time series, an adequate number of statistical tests are recommended to be applied so as to reach reliable conclusions for better decision-making [9]. Thus, the time series length must have at least 20–25 years length to ensure statistical validity of the trend results in hydrologic and meteorological variables [10].

2 Study Areas

In this study, seven (7) of the most important rivers of Greece were examined. These rivers were Acheloos, Aliakmonas, Axios, Evros, Nestos, Pinios and Strymonas. These rivers incorporate the largest catchments in Greece and have significant abstraction schemes and dams along their courses, while some of them have transboundary management issues as well (Axios, Strymonas, Nestos and Evros). The catchment sizes range from 6,240 km^2 for Nestos River to 53,560 km^2 for the Evros catchment (Table 1). Pinios and Strymonas rivers have catchment areas larger than 10,000 km^2 while Axios is the second largest catchment with an area of approximately 25,000 km^2. The main anthropogenic land cover in the study areas is agricultural land with Nestos having the smallest percentage (19% of the catchment) and Evros having the largest (56%), followed by Pinios (55%, Table 1) (Fig. 1).

Forest and seminatural areas range from 77% of the Nestos catchment to 39% in Evros, while only two out of seven catchments have a percentage below 50% (Evros and Pinios rivers; Table 1). The main water use in the particular river catchments is, therefore, irrigation which in many cases in FYROM reaches 73% of the total anthropogenic water consumption, while in Greece and in Greek and Turkish parts of the river basins, it reaches 90% (Table 2; [14–17]). On the contrary,

Table 1 Land cover areas distribution in the study catchments for the years 2006–2007

River	Catchment area	River length[a]	Agricultural areas		Artificial surfaces and bare land		Forest and seminatural areas		Waterbodies	
	km^2	km	km^2	%	km^2	%	km^2	%	km^2	%
Acheloos	6,468	255	1,508	23.3	342	5.3	4,336	67.0	282	4.4
Aliakmonas	8,874	310	3,725	42.0	149	1.7	4,843	54.6	157	1.8
Axios	24,596	380	10,356	42.1	498	2.0	13,606	55.3	136	0.6
Evros	53,597	550	29,855	55.7	2,222	4.1	20,963	39.1	557	1.0
Nestos	6,242	246	1,196	19.2	162	2.6	4,795	76.8	89	1.4
Pinios	10,739	257	5,887	54.8	276	2.6	4,548	42.4	28	0.3
Strymonas	17,063	410	6,268	36.7	620	3.6	10,001	58.6	175	1.0

Source: Liarikos et al. [11], European Environmental Agency [12]
[a][13]

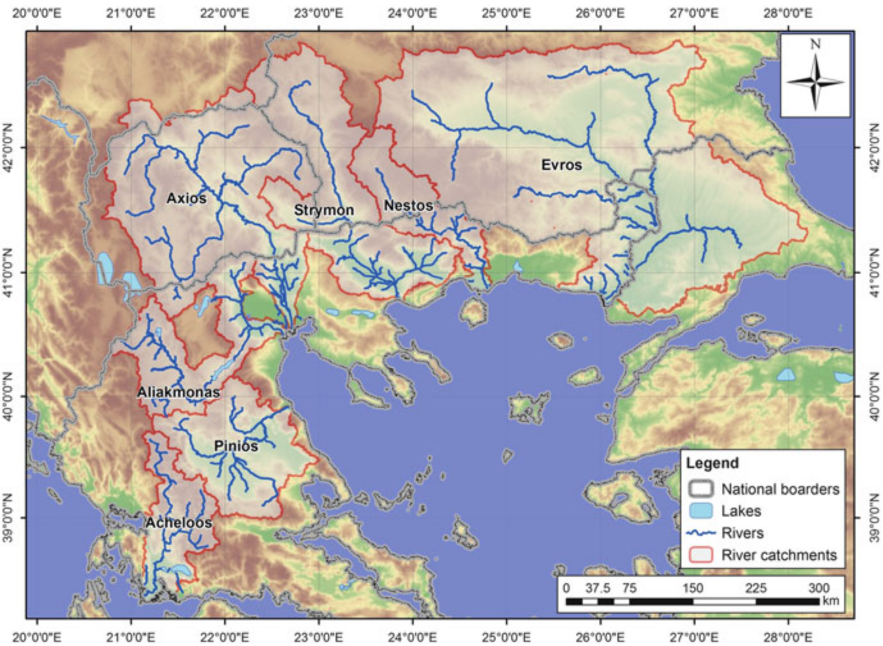


Fig. 1 Map with the study catchments and the hydrographic network

Table 2 Estimated water demand in river basins under study

River basin	Area		Country	River basin district (WD)	Area of WD	River basin area in WD	Irrigation[a]		Domestic use		Industrial water		Other		Total
	km^2	%			km^2		hm^3	%	hm^3	%	hm^3	%	hm^3	%	hm^3
Acheloos	6,468	100%	Greece	West Sterea Ellada (GR04)	10,399	100%	233.6	94%	13.9	6%	–	–	–	–	247.5
Aliakmonas	8,874	100%	Greece	West Macedonia (GR09)	13,624	100%	402.1	80%	28.5	6%	19.5	4%	52.1	10%	502.2
Axios	3,212	13%	Greece	Central Macedonia (GR10)	10,171	32%	169.1	75%	31.5	14%	25.3	11%	–	–	225.9
	20,511	83%	FYROM	Vardar River Basin	20,511	100%	731.7	55%	185.6	14%	233.0	18%	170.8	13%	1,321.2
	873	4%	Other	–	–	–	–	–	–	–	–	–	–	–	–
Total	24,596	100%	–	–	–	–	900.9	58%	217.1	14%	258.3	17%	170.8	11%	1,547.1
Evros	4,065	8%	Greece	Thrace (GR12)	11,195	36%	302.3	96%	10.1	3%	4.0	1%	–	–	316.4
	14,334	27%	Turkey	Meric-Ergene Basin (01)	14,560	100%	1,134.0	86%	112.0	8%	73.0	6%	–	–	1,319.0
	35,197	66%	Bulgaria	East Aegean River Basin District (BG3000)	35,197	100%	292.1	20%	70.8	5%	1,109.3	74%	18.7	1%	1,490.8
Total	53,597	100%	–	–	–	–	1,728.3	55%	192.9	6%	1,186.3	38%	18.7	1%	3,126.2
Nestos	2,799	45%	Greece	Thrace (GR12)	11,195	25%	208.1	96%	7.0	3%	2.8	1%	–	–	217.8
	3,443	55%	Bulgaria	West Aegean River Basin District (BG4000)	11,944	29%	3.4	16%	6.3	30%	8.8	42%	2.2	11%	20.7
Total	6,242	100%	–	–	–	–	211.5	89%	13.3	6%	11.6	5%	2.2	1%	238.6
Pinios	10,739	100%	Greece	Thessaly (GR08)	13,134	82%	1,278.0	96%	56.4	4%	–	–	–	–	1,334.4
Strymonas	6,228	37%	Greece	East Macedonia (GR11)	7,323	85%	538.2	95%	27.2	5%	–	–	–	–	565.4
	8,559	50%	Bulgaria	West Aegean River Basin District (BG4000)	11,944	72%	8.4	16%	15.7	30%	21.9	42%	5.6	11%	51.5
	1,527	9%	FYROM	Strumica River Basin	1,527	100%	117.9	73%	11.5	7%	32.9	20%	–	–	162.3
	749	4%	Other	–	–	–	–	–	–	–	–	–	–	–	–
Total	17,063	100%	–	–	–	–	664.5	85%	54.4	7%	54.8	7%	5.6	1%	779.3

[a]Livestock included (when available)

in Bulgaria, the main water consumption activity is mining and other forms of industry [16].

3 Data and Methodology

3.1 Precipitation Data

The meteorological data used in this study were obtained from various sources. More specifically, daily precipitation values were extracted from the Climate Prediction Center/National Centers for Environmental Prediction/National Weather Service/NOAA/US Department of Commerce [18]. The global summary of the day and month data set is obtained on a delayed monthly basis from the Climate Prediction Center (CPC) of the National Centers for Environmental Prediction (NCEP). CPC extracts surface synoptic weather observations from the Global Telecommunications System (GTS) and performs limited automated validation of the parameters. The data is then summarised for all reporting stations on a daily basis to current operational requirements related to the assessment of crop and energy production.

Moreover, precipitation measurements recorded at different time intervals were obtained from the Hellenic National Meteorological Service; the Ministry of Environment, Energy and Climate Change of Greece; the Republic Hydrometeorological Institute of FYROM; the National Institute of Meteorology and Hydrology of Bulgaria and the Turkish State Meteorological Service.

In all cases, the monthly precipitation amounts were compiled at each station, then for each catchment area the spatial monthly precipitation distribution was estimated based on the widely used Thiessen polygon deterministic approach, and afterwards the annual precipitation of the river basin for every hydrological year for the period 1980/1981–2008/2009 was calculated.

3.2 Discharge Data

Due to the lack of satisfactory discharge time series at the downstream parts of each catchment examined, the results from E-HYPE pan-European hydrological model were used (European – HYdrological Predictions for the Environment). E-HYPE 2.1 is a model application for the entire European continent whereby hydrological flows and nutrient processes are calculated daily for each class within a subbasin level. Calculations are made in subbasins with a median size of 215 km^2. The model aims to take into account important processes including both hydrological and anthropogenic impacts for all regions across Europe (e.g. irrigation, hydropower). The results delivered for each subbasin in E-HypeWeb are discharge (Q, m^3/s) and monthly total loads of nitrogen (TN, kg/month) and phosphorus (TP, kg/month) (including the contributions of discharge and nutrients from all subbasins upstream of the chosen subbasin). The accuracy of results is improved considerably for catchments 7,000 km^2 or more in upstream area as this is the resolution of the input forcing data [19, 20].

So far the E-HYPE hydrological model results have not been validated against river discharge observations in southeastern Europe and Greece. Therefore, the comparison between average monthly discharge time series from the E-HYPE model of each subbasin examined and in situ observation was considered necessary. The behaviour and the performance of the model were examined with efficiency criteria, which are defined as quantitative measure of performance, goodness of fit or likelihood [21]. For scientific sound model validation, a combination of different efficiency criteria complemented by the assessment of the absolute or relative volume error is recommended [22].

In this study, the criteria used to check the model reliability were the following: mean error (ME), mean absolute error (MAE), mean absolute percentage error (MAPE), root mean squared error (RMSE), Pearson's correlation coefficient (R), squared correlation coefficient (R^2) and Nash–Sutcliffe coefficient of efficiency (Nr). The efficiency criteria relative error (RE) contains a summation of the error term (difference between observations and simulated time series at each time step) normalised by a measure of the variability in the observations. In order to avoid the cancelling of errors of opposite sign, the summation of the mean absolute (MAE) or root mean squared (RMSE) errors is often used [22].

Singh et al. [23] state that RMSE and MAE values less than half the standard deviation of the measured data may be considered low. The Nash–Sutcliffe coefficient of efficiency is commonly used in hydrological modelling and determines the relative magnitude of the residual variance ("noise") compared to the measured data variance ("information") [24]. Generally, due to the fact that correlation and correlation-based measures are oversensitive to extreme values (outliers) and are insensitive to additive and proportional differences between model predictions and observations, it is recommended that additional evaluation measures (such as summary statistics and absolute error measures) supplement hydrological model evaluation [25].

3.3 Statistical Analysis

3.3.1 Normality

Prior to applying any statistical tests to the time series, normality of the annual data was tested. In order to examine the normality of the annual precipitation and discharge time series, three (3) statistical tests were employed (Kolmogorov–Smirnov, KS; Lilliefors, LF; and Shapiro–Wilk, SW). The Kolmogorov–Smirnov (KS) test [26] is an empirical distribution function (EDF) which finds the difference between cumulative distribution of the time series data and the expected cumulative normal distribution and computes its p-value for the largest. The Lilliefors (LF) test [27] is a modification of the Kolmogorov–Smirnov test and compares the cumulative distribution of data to the expected cumulative normal distribution. The Shapiro–Wilk (SW) test [28], which is one of the most powerful normality tests, is similar to computing a correlation between the quantiles of the standard normal distribution and the ordered data points of a hydrologic time series [29, 30].

3.3.2 Homogeneity

Homogeneity or consistency implies that the hydrologic time series data belong to the same statistical population having a time invariant mean. Non-homogeneity arises due to changes in the method of data collection and/or the environment in which it is done [31]. Therefore, the homogeneity or consistency tests are based on evaluating the significance of changes in the mean value [29].

In order to examine the homogeneity of the annual precipitation and discharge time series, three statistical tests were employed (von Neumann test, cumulative deviations test and Bayesian test). The von Neumann ratio (N) is defined as the ratio of the mean square successive (year to year) difference to the variance and has an expected value of 2 for a homogeneous series, but it tends to be less than 2 for a non-homogeneous series [32]. The cumulative deviations test is based on the adjusted partial sums or cumulative deviations from the mean [33]. The Bayesian test was developed by Chernoff and Zacks [34] and was modified later by Gardner Jr [35].

3.3.3 Stationarity

A time series of hydrological data is strictly stationary, if its statistical properties (e.g. its mean, variance and higher-order moments) are unaffected by the choice of time origin. In this case, the form of stationarity was identified based on tests concerning the stability of the variance and the stability of the mean. Although stability of these two properties indicates only a weak form of stationarity, this is enough to identify a nonstationary time series [36].

In order to apply the parametric t-test and non-parametric Mann–Whitney test for stability of mean of the time series of discharge and rainfall, each time series was divided into two subseries and three subseries. Then the stationary tests were applied for various combinations of the subseries in order to examine whether the means are significantly different [29, 36, 37].

3.3.4 Trend

A common deterministic component in a time series is a trend. A trend is a tendency for successive values to be increasing or decreasing over time [38]. Changes in hydrologic conditions by natural and/or artificial factors can introduce linear or non-linear trends into a hydrologic time series. The trend in a time series can be expressed by a suitable linear or non-linear model; the linear model is widely used in hydrology [39].

Various tests have been reported for detecting trend in a hydrologic time series. In this effort, due to, in some cases, non-normal distribution of the time series, the rank-based non-parametric Mann–Kendall [40, 41] and Spearman's (Rho) tests for trend were applied [42–44]; these tests have been commonly used to assess the significance of trends in hydro-meteorological time series such as river discharge and precipitation [45].

The sequential version of the Mann–Kendall (SMK) test allows detection of approximate change of trend with time [43]. The distribution-free CUSUM test is a non-parametric rank-based method that tests whether the means in two parts of a record are different for an unknown time of change. In particular, successive observations are compared with the median of the series in order to detect a change in the mean of a time series after a number of observations [46]. Finally, the trend magnitude was estimated based on Sen's estimator of slope. This non-parametric statistic is applied in cases of linear trend and determines the magnitude of change per unit time [47]. Sen's test for the estimation of slope requires a time series of equally spaced data [29].

4 Results

4.1 Precipitation Data

For the particular study, 61 different meteorological stations have been used to estimate the amount of precipitation falling in the seven different catchments during the past 3 decades (approximately 9 stations per catchment, Table 3). The Thiessen polygon weights and area of influence for each station have also been estimated and used in the aforementioned estimations (Table 3, Fig. 2).

4.2 Discharge Data

At Table 4, the statistical characteristics and the efficiency criteria for the validation of the hydrological model E-HYPE compared to measured discharge are presented. Based on the results, in all cases the correlation coefficient R was characterised as

Table 3 Meteorological stations used for the calculation of spatial precipitation

River catchment	a/a	WMO	Name	Country	Owner	River basin	Latitude WGS84 (dd)	Longitude	Elevation (m)	Area affected (km^2)	Station weight
Acheloos	1	16672	Agrinio	Greece	HNMS	Acheloos	38.606	21.353	24.0	1,458.6	0.23
	2	–	Granitsa	Greece	MEECC	Acheloos	39.104	21.509	905.7	1,387.4	0.21
	3	–	Karpenisi	Greece	MEECC	Acheloos	38.915	21.793	962.2	1,165.2	0.18
	4	–	Katafyto	Greece	MEECC	Acheloos	39.635	21.246	1,018.4	881.5	0.14
	5	–	Perdikaki	Greece	MEECC	Acheloos	39.050	21.362	749.8	748.1	0.12
	6	–	Stanos	Greece	MEECC	Acheloos	38.805	21.181	33.1	551.8	0.09
	7	–	Trikorfo	Greece	MEECC	Evinos	38.411	21.640	100.1	275.3	0.04
Total										6,481.0	1.00
Aliakmonas	1	–	Agiofyllo	Greece	MEECC	Pinios	39.864	21.562	584.1	913.2	0.10
	2	13583	Bitola	FYROM	YXMP	Axios	41.050	21.367	5,890.0	564.0	0.06
	3	–	Giannota	Greece	MEECC	Pinios	39.981	22.040	554.0	460.6	0.05
	4	–	Grevenitiko	Greece	MEECC	Arachtos	39.806	21.005	1,005.6	244.6	0.03
	5	–	Kastanea Imathias	Greece	MEECC	Aliakmonas	40.408	22.121	1,053.4	1,781.8	0.20
	6	16614	Kastoria	Greece	HNMS	Aliakmonas	40.450	21.280	604.0	1,763.5	0.20
	7	–	Katerini	Greece	MEECC	Other	40.277	22.513	30.4	18.8	0.00
	8	–	Siatista	Greece	MEECC	Aliakmonas	40.262	21.551	933.1	989.6	0.11
	9	16619	Trikala Imathias	Greece	HNMS	Loudias	40.600	22.550	5.9	1,056.1	0.12
	10	–	Tsotilio	Greece	MEECC	Aliakmonas	40.260	21.325	856.0	1,081.9	0.12
Total										8,882.1	1.00
Axios	1	13583	Bitola	FYROM	YXMP	Axios	41.050	21.367	5,890.0	4,601.3	0.19
	2	13592	Demir Kapija	FYROM	YXMP	Axios	41.417	22.250	1,260.0	3,202.0	0.13
	3		Goumenissa	Greece	MEECC	Axios	40.938	22.459	212.0	1,319.7	0.05
	4	–	Kilkis	Greece	MEECC	Gallikos	40.992	22.884	261.5	987.6	0.04

	5	13493	Kriva Palanka	FYROM	YXMP	Axios	42.200	22.333	6,960.0	2,787.9	0.11
	6	13578	Ohrid	FYROM	YXMP	Drin	41.117	20.800	7,610.0	1,284.9	0.05
	7	13586	Skopje airport	FYROM	YXMP	Axios	41.967	21.650	2,390.0	7,119.4	0.29
	8	13591	Stip	FYROM	YXMP	Axios	41.750	22.183	3,270.0	3,131.9	0.13
	9	16622	Thessaloniki airport	Greece	HNMS	Axios	40.517	22.967	40.0	161.4	0.01
Total										24,605.3	1.00
Evros	1	15615	Mussala	Bulgaria	NIMH	Evros	42.170	23.580	2,927.0	4,520.7	0.08
	2	15640	Silven	Bulgaria	NIMH	Evros	42.680	26.270	222.0	7,312.4	0.14
	3	15730	Kardjali	Bulgaria	NIMH	Evros	41.630	25.400	243.0	8,129.0	0.15
	4	17050	Edirne	Turkey	TSMS	Evros	41.670	26.570	48.0	8,191.2	0.15
	5	16627	Alexandroupoli airport	Greece	HNMS	Evros	40.850	25.920	3.0	2,324.8	0.04
	6	17056	Tekirdag	Turkey	TSMS	Evros	40.980	27.480	4.0	6,311.2	0.12
	7	15627	Botev Vrah	Bulgaria	NIMH	Evros	42.720	24.920	2,389.0	7,430.8	0.14
	8	–	Sitochori	Greece	MEECC	Evros	41.461	26.354	130.7	3,522.8	0.07
	9	–	Megalo Dereio	Greece	MEECC	Evros	41.232	26.022	381.6	2,402.9	0.04
	10	–	Mikrokleisoura	Greece	MEECC	Nestos	41.387	24.057	457.4	1,655.9	0.03
	11	–	Echinos	Greece	MEECC	Xiropotamos	41.278	24.970	329.8	1,795.3	0.03
Total										53,617.4	1.00
Nestos	1	15615	Mussala	Bulgaria	NIMH	Evros	42.170	23.580	2,927.0	1,160.1	0.19
	2	–	Mikrokleisoura	Greece	MEECC	Nestos	41.387	24.057	457.4	2,168.0	0.35
	3	–	Chrysoupoli	Greece	MEECC	Nestos	40.985	24.707	20.4	935.3	0.15
	4	–	Paranesti	Greece	MEECC	Nestos	41.267	24.500	122.4	1,271.6	0.20
	5	15712	Sandanski	Bulgaria	NIMH	Strymonas	41.570	23.280	191.0	707.0	0.11
Total										6,390.2	1.00

(continued)

Table 3 (continued)

River catchment	a/a	WMO	Name	Country	Owner	River basin	Latitude WGS84 (dd)	Longitude	Elevation (m)	Area affected (km^2)	Station weight
Pinios	1	–	Agiofyllo	Greece	MEECC	Pinios	39.864	21.562	584.1	1,004.5	0.09
	2	–	Farkadona	Greece	MEECC	Pinios	39.588	22.070	86.2	1,345.1	0.13
	3	–	Giannota	Greece	MEECC	Pinios	39.981	22.040	554.0	1,484.1	0.14
	4	16648	Larissa	Greece	HNMS	Pinios	39.630	22.450	73.0	1,404.7	0.13
	5	–	Rentina	Greece	MEECC	Pinios	39.066	21.974	884.9	946.7	0.09
	6	–	Skopia	Greece	MEECC	Pinios	39.155	22.468	444.7	1,127.4	0.10
	7	–	Sotirio	Greece	MEECC	Pinios	39.505	22.706	52.4	882.3	0.08
	8	16645	Trikala	Greece	HNMS	Pinios	39.550	21.767	108.6	1,642.1	0.15
	9	–	Zappio	Greece	MEECC	Pinios	39.462	22.440	172.3	902.1	0.08
Total										10,741.3	1.00
Strymonas	1	–	Aidonochori	Greece	MEECC	Strymonas	40.844	23.723	186.3	1,502.4	0.09
	2	–	Alistrati	Greece	MEECC	Strymonas	41.061	23.959	281.4	3,006.2	0.18
	3	13592	Demir Kapija	FYROM	YXMP	Axios	41.417	22.250	125.0	685.3	0.04
	4	–	Kilkis	Greece	MEECC	Gallikos	40.992	22.884	261.5	967.5	0.06
	5	13493	Kriva Palanka	FYROM	YXMP	Axios	42.200	22.333	6,960.0	2,728.0	0.16
	6	15615	Mussala	Bulgaria	NIMH	Evros	42.170	23.580	2,927.0	1,882.7	0.11
	7	15712	Sandanski	Bulgaria	NIMH	Strymonas	41.570	23.280	191.0	4,266.4	0.25
	8	15614	Sofia	Bulgaria	NIMH	Danube	42.820	23.380	595.0	1,275.0	0.07
	9	13591	Stip	FYROM	YXMP	Axios	41.750	22.180	326.0	360.6	0.02
	10	13489	Vranje	FYROM	YXMP	Danube	42.550	21.920	433.0	389.0	0.02
Total										17,075.7	1.00

HNMS Hellenic National Meteorological Service; *YXMP* Republic Hydrometeorological Institute of FYROM; *MEECC* Ministry of Environment, Energy and Climate Change of Greece; *NIMH* National Institute of Meteorology and Hydrology of Bulgaria; *TSMS* Turkish State Meteorological Service

Fig. 2 Thiessen polygons and meteorological stations' locations

moderate (e.g. Axios River, 0.58) to strong (e.g. Evros, 0.83), indicating the relatively sufficient performance of the model. Squared correlation coefficient R^2 was characterised in all cases as moderate to high and acceptable [48], with the exception of Axios River where the respective value is smaller than 0.5. The Nash–Sutcliffe coefficient Nr was in most cases greater than zero, with the exception of Aliakmonas and Axios rivers, indicating that for these rivers the mean observed value is a better predictor than the simulated value. Overall, Nr values between 0.0 and 1.0 are generally viewed as acceptable levels of performance [48] (Fig. 3).

It must be noted that by definition the sum of squared errors (RMSE) and the modelling efficiency (Nash–Sutcliffe coefficient) are sensitive to extreme values (outliers) and timing errors in the predictions [21]. Taking under consideration that the observed discharge data of Greek rivers, especially prior to the year 2000, were obtained manually by scientific or technical personnel in situ, some inaccuracies especially during high flows should be expected. Therefore, the negative values of Nr could be partially attributed to the subjective assessment and possible incorrect measurements of extreme discharge values.

Table 4 Statistical characteristics and the efficiency criteria for the validation of the hydrological model E-HYPE

River	N	ME	MAE	MAPE	RMSE	R	R2	Nr	Measurement site
Acheloos	348	−47.4	52.3	44.1	76.0	0.82	0.67	0.41	Kastraki dam
Aliakmonas	312	29.2	39.4	110.6	59.0	0.78	0.61	−1.44	Polyfyto dam
Axios	136	83.5	113.9	234.9	169.3	0.58	0.34	−1.52	Chalastra
Evros	272	−113.1	131.8	47.2	198.1	0.83	0.69	0.50	Pythio
Nestos	201	6.0	16.1	65.7	26.2	0.76	0.58	0.23	Temenos
Pinios	89	−18.5	34.1	61.8	50.1	0.78	0.61	0.46	Amygdalea
Strymonas	44	−2.2	47.1	107.5	63.7	0.81	0.65	0.61	Mirkino

N number of observations, *ME* mean error, *MAE* mean absolute error, *MAPE* mean absolute percentage error, *RMSE* root mean square error, *R* correlation coefficient, R^2 squared correlation coefficient, *Nr* Nash–Sutcliffe coefficient

4.3 *Statistical Analysis*

4.3.1 Descriptive Statistics

In this case study, annual discharge and precipitation data for the hydrological years 1980/1981–2008/2009 have been analysed and the descriptive statistics of these time series appears in Table 5. The highest average annual precipitation was recorded in the Acheloos catchment (1,096 mm) and the lowest respective value in the Axios catchment (493 mm). For discharge, the highest average annual value was observed in Evros River (250 m^3/s) and the lowest in Nestos River (53 m^3/s, Table 5).

The highest variations in the precipitation values appear in the Acheloos catchment, while the lowest ones in Evros, and similarly for the discharge, the highest variations appear in Evros River, while the lowest ones in Nestos (Fig. 4).

The scatter diagrams between precipitation and discharge indicate a fairly strong correlation between the two variables for most of the studied rivers. The correlation coefficients fluctuate from 0.48 in Nestos River up to 0.82 in Pinios and 0.80 in Aliakmonas rivers. The high correlation coefficients mean that precipitation explains most of the interannual variability of discharge in the particular catchments while where the aforementioned coefficients are low, other factors mostly determine the average annual discharge (e.g. abstractions, evaporation, etc.). The relationship between precipitation and discharge is well described from the identified least squares equations in Aliakmonas and Pinios rivers, while the catchment with the lowest predictability of discharge based on precipitation is Nestos River followed by Evros (Fig. 5).

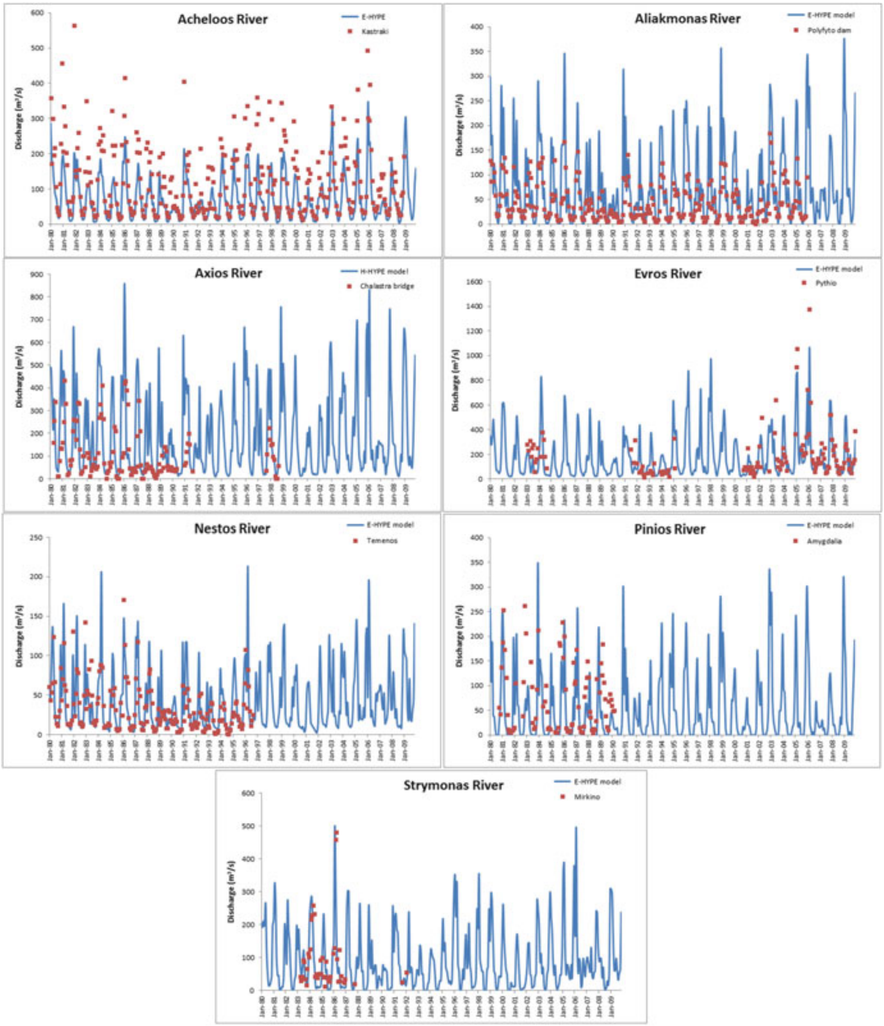


Fig. 3 Discharge modelled fluctuations and respective measurements during the last 3 decades

4.3.2 Normality

Based on the Kolmogorov–Smirnov (KS) normality test applied, the time series examined can be considered normal, since the *p*-value is more than 0.05 in all cases and therefore the null hypothesis of normality cannot be rejected. Likewise, the results of Shapiro–Wilk (SW) normality test indicated the presence of normality in

Table 5 Descriptive statistics for the annual time series of discharge and precipitation for all rivers

River/ param.[a]	Valid *N*	Mean	Median	Minimum	Maximum	Variance	Std. dev.	Coef. var.
Pach	29	1,095.50	1,097.20	743.9	1,407.50	35,801.90	189.2	17.3
Pal	29	646.3	663	427.2	889.2	16,503.80	128.5	19.9
Pax	29	492.5	522.5	223.8	673.2	12,627.20	112.4	22.8
Pev	29	637.6	619.1	498.6	832.5	9,938.40	99.7	15.6
Pnes	29	632.6	621.5	460.5	986.3	14,543.90	120.6	19.1
Ppin	29	610.9	627.5	394.5	836.8	14,881.20	122	20
Pstr	29	562.9	582.1	366.2	733.4	9,342.00	96.7	17.2
Dach	29	72.5	72.2	32.2	111.3	503.1	22.4	30.9
Dal	29	72.9	70.2	32.5	112.3	537.9	23.2	31.8
Dax	29	185.7	176.9	89.1	269.5	2,889.70	53.8	28.9
Dev	29	249.7	247.2	86.5	448.7	9,455.20	97.2	38.9
Dnes	29	52.6	50.9	26	84.7	216.5	14.7	28
Dpin	29	74.3	72.5	24	135.9	952.3	30.9	41.6
Dstr	29	87.4	89.9	37.6	146	824.3	28.7	32.9

[a]The first letter refers to the parameter (*P* precipitation, *D* discharge) and the rest of the letters to the river name (*ach* Acheloos, *al* Aliakmonas, *ax* Axios, *ev* Evros, *nes* Nestos, *pin* Pinios, *str* Strymonas)

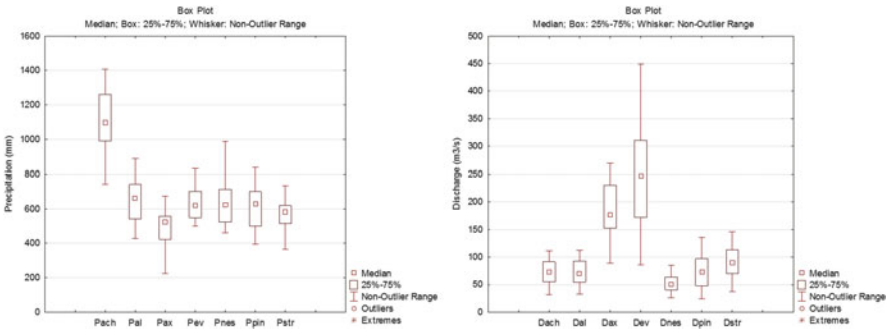


Fig. 4 Box plots of precipitation and discharge in each study river

annual precipitation and river discharge of the time series examined (Table 6). Only in two cases the *p*-value is smaller than 0.05 at Lilliefors normality test for the annual precipitation of Axios and Strymonas rivers, which seems to be attributed to some extremely wet hydrological years.

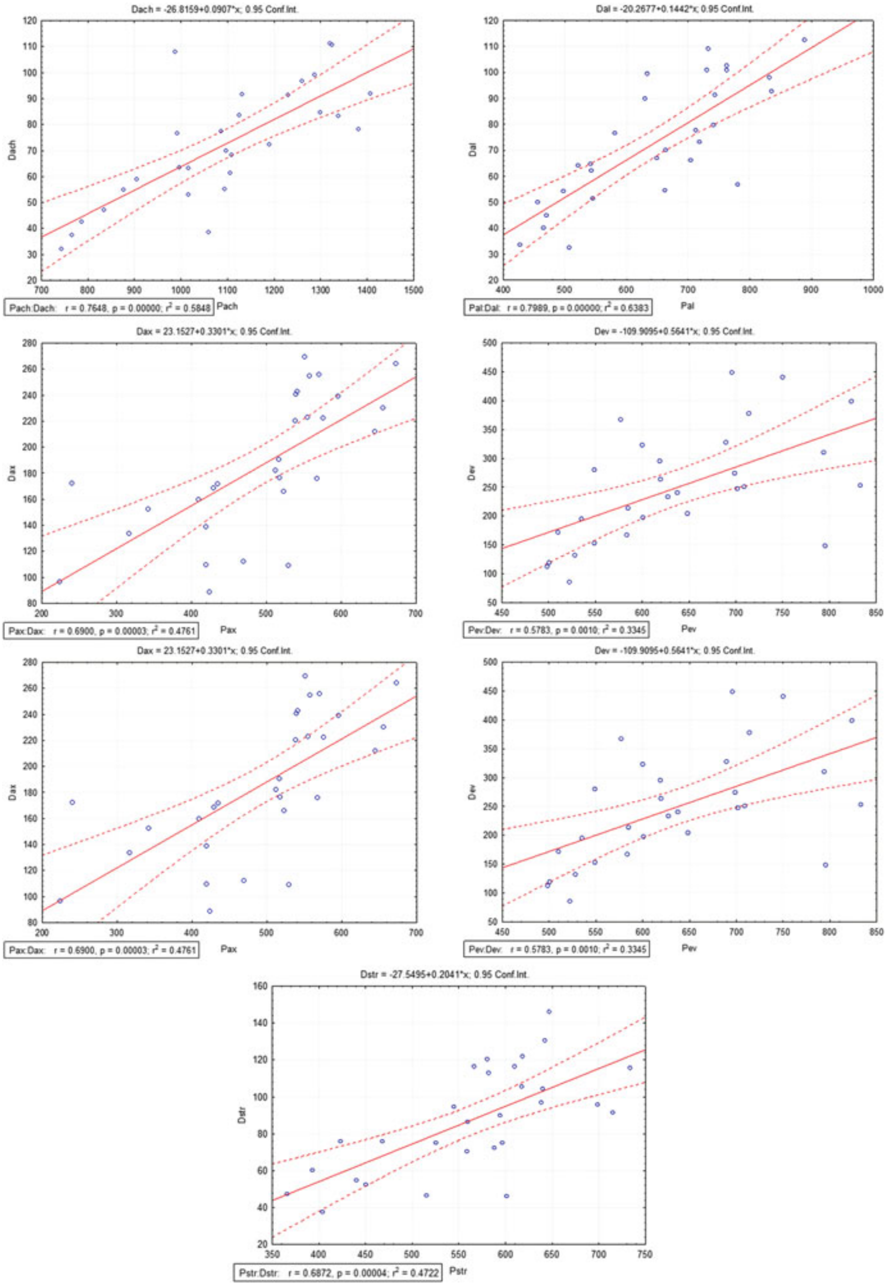


Fig. 5 Scatter diagrams between precipitation and discharge for each river basin

Table 6 Results of normality tests

River/ param.	Kolmogorov–Smirnov (KS) Test statistic	*p*-value	Normal	Lilliefors (LF) *p*-value	Normal	Shapiro–Wilk (SW) Test statistic	*p*-value	Normal
Pach	0.087	***p*>0.20**	**Yes**	***p* > 0.20**	**Yes**	0.962	0.37	Yes
Dach	0.076	***p*>0.20**	**Yes**	***p*>0.20**	**Yes**	0.973	0.65	Yes
Pal	0.129	***p*>0.20**	**Yes**	***p*>0.20**	**Yes**	0.957	0.28	Yes
Dal	0.111	***p*>0.20**	**Yes**	***p*>0.20**	**Yes**	0.960	0.33	Yes
Pax	0.189	***p*>0.20**	**Yes**	*p*<0.01	No	0.933	0.07	Yes
Dax	0.121	***p*>0.20**	**Yes**	***p*>0.20**	**Yes**	0.952	0.20	Yes
Pev	0.092	***p*>0.20**	**Yes**	***p*>0.20**	**Yes**	0.944	0.13	Yes
Dev	0.072	***p*>0.20**	**Yes**	***p*>0.20**	**Yes**	0.972	0.62	Yes
Pnes	0.094	***p*>0.20**	**Yes**	***p*>0.20**	**Yes**	0.943	0.12	Yes
Dnes	0.107	***p*>0.20**	**Yes**	***p*>0.20**	**Yes**	0.975	0.69	Yes
Ppin	0.126	***p*>0.20**	**Yes**	***p*>0.20**	**Yes**	0.953	0.22	Yes
Dpin	0.122	***p*>0.20**	**Yes**	***p*>0.20**	**Yes**	0.967	0.49	Yes
Pstr	0.139	***p*>0.20**	**Yes**	*p*<0.15	No	0.955	0.24	Yes
Dstr	0.105	***p*>0.20**	**Yes**	***p*>0.20**	**Yes**	0.970	0.55	Yes

The results in bold indicate which series have normal distributions for a 5% significance level

Table 7 Results of homogeneity tests

Time series	von Neumann *N*	Cumulative deviations *Q*/sqt(*n*)	*R*/sqt(*n*)	Bayesian *U*	*A*
Pach	**2.069**	**0.744**	**0.964**	**0.154**	**1.156**
Dach	**2.161**	**0.691**	**0.971**	**0.119**	**0.738**
Pal	**2.475**	**0.538**	**0.905**	**0.081**	**0.481**
Dal	**2.208**	**0.866**	**1.275**	**0.101**	**0.652**
Pax	**2.003**	**0.901**	**1.398**	**0.177**	**1.291**
Dax	**2.110**	**0.659**	**1.181**	**0.093**	**0.667**
Pev	**1.633**	**0.866**	**1.315**	**0.113**	**0.908**
Dev	**1.638**	1.254	**1.410**	**0.332**	**1.564**
Pnes	**1.491**	**1.200**	1.597	**0.363**	**2.354**
Dnes	**1.516**	**1.102**	**1.437**	**0.312**	**1.765**
Ppin	**2.052**	**0.824**	**1.028**	**0.124**	**0.919**
Dpin	**2.416**	**0.428**	**0.731**	**0.044**	**0.294**
Pstr	1.241	**1.189**	**1.769**	**0.380**	**2.070**
Dstr	**1.800**	**0.866**	**1.275**	**0.175**	**1.080**
Critical values (5%)	1.461	1.240	1.500	0.444	2.420

The results in bold indicate which series are homogeneous at 5% significance level

4.3.3 Homogeneity

Based on the results of the von Neumann test, the time series examined are homogeneous, since in all cases (except the precipitation of Strymonas River) the N ratio is smaller than the critical value at 5% significance level.

In the cases of cumulative deviations and Bayesian tests, all the applied test statistics (Q, R, U and A) have smaller values compared to their critical values [33] at 5% significance level. Particularly, this means that the annual rainfall and discharge series are homogeneous and belong to populations with similar characteristics. The only exceptions are noted in the cases of Evros River annual discharge and Nestos River annual precipitation at cumulative deviations test, where the test statistics slightly reach the critical values (Table 7). Based on the results, some non-homogeneity issues arise, which could be attributed to changes in the method of data collection and/or the environment in which it is done [31].

4.3.4 Stationarity

Based on the results of the *t*-test and Mann–Whitney *U* test applied on five subseries of precipitation and discharge data of the major river catchments of Greece, in most cases the values of the statistical tests are less than their critical values, and thus, the null hypothesis cannot be rejected at 5% significance level, and the annual discharge and precipitation time series can be considered stationary (Tables 8 and 9).

Only in the cases of Evros River discharge and only for one pair of subseries (1980/1981–1994/1994 and 1995/1996–2008/2009), Nestos River precipitation (subseries 1980/1981–1989/1990 and 1990/1991–1999/2000, 1990/1991–1999/2000 and 2000/2001–2008/2009) and Strymonas River precipitation (subseries 1990/1991–1999/2000 and 2000/2001–2008/2009), the absolute value of the *t*-value computed from the *t*-test was higher than the *t*-critical at 5% significant level, while also the *z*-value from the Mann–Whitney test was higher than the critical *z*-value (± 1.96) and the *U*-value was smaller than the *U*-critical; therefore, the null hypothesis (existence of stationarity) at 5% significant level in these cases cannot be concluded. This practically means that there is a relatively strong trend in these time series that alters their basic statistical characteristics (mean and variance) through time (Tables 8 and 9).

4.3.5 Trend

Based on the results of various trend detection tests, no significant, single trend at the 5% level of the overall annual precipitation and discharge was detected, for the period 1980/1981–2008/2009 (Table 10). This means that there are various trends that change their characteristics throughout the study period but not a single trend for the entire time period.

Table 8 Statistical characteristics of the subseries examined

Subseries	Statistics	Pach	Dach	Pal	Dal	Pax	Dax	Pev	Dev	Pnes	Dnes	Ppin	Dpin	Pstr	Dstr
1980/1981–1994/1994	*N*	15	15	15	15	15	15	15	15	15	15	15	15	15	15
	Mean	1,105.7	68.1	632.4	68.9	484.5	178.3	631.9	210.0	640.1	47.7	615.8	71.6	533.2	80.1
	StDev	219.6	20.3	137.1	20.5	129.9	50.1	114.5	69.8	145.3	12.8	138.0	27.6	100.7	26.6
	Var	48,233.1	411.0	18,792.3	422.1	16,863.3	2,507.0	13,121.6	4,871.6	21,118.2	163.7	19,036.9	759.8	10,147.7	709.8
1995/1996–2008/2009	*N*	14	14	14	14	14	14	14	14	14	14	14	14	14	14
	Mean	1,084.5	77.2	661.2	77.3	501.2	193.7	643.6	292.3	624.6	57.9	605.6	77.1	594.8	95.1
	StDev	157.9	24.4	121.8	25.8	94.2	58.2	84.8	106.5	91.9	15.2	107.2	34.9	84.1	29.8
	Var	24,917.7	595.6	14,845.4	665.0	8,881.1	3,390.8	7,199.4	11,351.3	8,449.0	231.0	11,493.6	1,215.6	7,078.4	887.1
1980/1981–1989/1990	*N*	10	10	10	10	10	10	10	10	10	10	10	10	10	10
	Mean	1,126.6	68.5	629.4	69.5	501.7	185.0	639.7	213.5	694.2	50.1	618.8	70.4	557.6	85.7
	StDev	211.9	20.9	129.2	21.9	147.1	51.7	115.2	71.4	143.9	12.9	142.5	29.2	98.7	25.4
	Var	44,888.9	436.7	16,701.0	478.1	21,639.5	2,676.0	13,260.3	5,094.8	20,696.5	166.4	20,298.4	851.6	9,735.7	644.6
1990/1991–1999/2000	*N*	10	10	10	10	10	10	10	10	10	10	10	10	10	10
	Mean	1,102.7	72.9	639.8	73.7	460.2	184.7	627.6	266.2	557.4	50.5	612.4	77.5	511.1	83.8
	StDev	184.5	18.3	133.1	19.9	113.0	47.2	103.7	105.4	88.1	14.0	118.3	29.7	93.0	29.4
	Var	34,026.9	334.2	17,719.5	395.1	12,780.2	2,224.9	10,761.4	11,118.8	7,762.4	195.1	13,999.7	882.1	8,642.6	863.4
2000/2001–2008/2009	*N*	9	9	9	9	9	9	9	9	9	9	9	9	9	9
	Mean	1,052.9	76.5	672.2	75.9	518.3	187.7	646.2	271.8	647.9	57.8	600.4	74.9	626.4	93.2
	StDev	182.2	29.3	133.7	29.6	57.5	67.8	86.9	111.0	82.6	17.5	115.3	36.7	63.9	33.6
	Var	33,209.8	856.0	17,873.0	875.0	3,311.1	4,593.8	7,546.8	12,327.7	6,818.6	307.8	13,293.0	1,350.2	4,086.2	1,131.6

Table 9 Comparison of the results of t-test and Mann–Whitney test for various combinations of discharge and precipitation subseries

x	Subseries 1	Subseries 2	t-test				Mann–Whitney U test				
			t	$P(T<=t)$ two tail	t-critical two tail	Significance	U	z	p-value	U-critical	Significance
Pach	**1980/1981–1994/1994**	**1995/1996–2008/2009**	**0.297**	**0.769**	**2.052**	**No**	**97**	**0.327**	**0.743**	**59**	**No**
	1980/1981–1989/1990	**1990/1991–1999/2000**	**0.269**	**0.791**	**2.101**	**No**	**47**	**0.189**	**0.850**	**23**	**No**
	1980/1981–1989/1990	**2000/2001–2008/2009**	**0.808**	**0.430**	**2.110**	**No**	**34**	**0.857**	**0.391**	**20**	**No**
	1990/1991–1999/2000	**2000/2001–2008/2009**	**0.591**	**0.562**	**2.110**	**No**	**34**	**0.857**	**0.391**	**20**	**No**
Dach	**1980/1981–1994/1994**	**1995/1996–2008/2009**	**−1.086**	**0.287**	**2.052**	**No**	**82**	**−0.982**	**0.326**	**59**	**No**
	1980/1981–1989/1990	**1990/1991–1999/2000**	**−0.493**	**0.628**	**2.101**	**No**	**45**	**−0.340**	**0.734**	**23**	**No**
	1980/1981–1989/1990	**2000/2001–2008/2009**	**−0.686**	**0.502**	**2.110**	**No**	**36**	**−0.694**	**0.488**	**20**	**No**
	1990/1991–1999/2000	**2000/2001–2008/2009**	**−0.326**	**0.748**	**2.110**	**No**	**43**	**−0.122**	**0.903**	**20**	**No**
Pal	**1980/1981–1994/1994**	**1995/1996–2008/2009**	**−0.597**	**0.555**	**2.052**	**No**	**92**	**−0.546**	**0.585**	**59**	**No**
	1980/1981–1989/1990	**1990/1991–1999/2000**	**−0.179**	**0.860**	**2.101**	**No**	**49**	**−0.038**	**0.970**	**23**	**No**
	1980/1981–1989/1990	**2000/2001–2008/2009**	**−0.710**	**0.487**	**2.110**	**No**	**37**	**−0.612**	**0.540**	**20**	**No**
	1990/1991–1999/2000	**2000/2001–2008/2009**	**−0.529**	**0.604**	**2.110**	**No**	**39**	**−0.449**	**0.653**	**20**	**No**
Dal	**1980/1981–1994/1994**	**1995/1996–2008/2009**	**−0.971**	**0.340**	**2.052**	**No**	**80**	**−1.069**	**0.285**	**59**	**No**
	1980/1981–1989/1990	**1990/1991–1999/2000**	**−0.448**	**0.660**	**2.101**	**No**	**43**	**−0.491**	**0.623**	**23**	**No**

(continued)

Table 9 (continued)

x	Subseries 1	Subseries 2	*t*-test				Mann–Whitney *U* test				
			t	$P(T<=t)$ two tail	*t*-critical two tail	Significance	U	z	*p*-value	*U*-critical	Significance
	1980/1981–1989/1990	**2000/2001–2008/2009**	**−0.541**	**0.596**	**2.110**	**No**	**37**	**−0.612**	**0.540**	**20**	**No**
	1990/1991–1999/2000	**2000/2001–2008/2009**	**−0.194**	**0.848**	**2.110**	**No**	**42**	**−0.204**	**0.838**	**20**	**No**
Pax	**1980/1981–1994/1994**	**1995/1996–2008/2009**	**−0.394**	**0.697**	**2.052**	**No**	**92**	**−0.546**	**0.585**	**59**	**No**
	1980/1981–1989/1990	**1990/1991–1999/2000**	**0.708**	**0.488**	**2.101**	**No**	**39**	**0.794**	**0.427**	**23**	**No**
	1980/1981–1989/1990	**2000/2001–2008/2009**	**−0.317**	**0.755**	**2.110**	**No**	**45**	**0.041**	**0.967**	**20**	**No**
	1990/1991–1999/2000	**2000/2001–2008/2009**	**−1.387**	**0.183**	**2.110**	**No**	**31**	**−1.102**	**0.270**	**20**	**No**
Dax	**1980/1981–1994/1994**	**1995/1996–2008/2009**	**−0.768**	**0.449**	**2.052**	**No**	**84**	**−0.895**	**0.371**	**59**	**No**
	1980/1981–1989/1990	**1990/1991–1999/2000**	**0.016**	**0.987**	**2.101**	**No**	**48**	**−0.113**	**0.910**	**23**	**No**
	1980/1981–1989/1990	**2000/2001–2008/2009**	**−0.098**	**0.923**	**2.110**	**No**	**42**	**−0.204**	**0.838**	**20**	**No**
	1990/1991–1999/2000	**2000/2001–2008/2009**	**−0.114**	**0.910**	**2.110**	**No**	**43**	**−0.122**	**0.903**	**20**	**No**
Pev	**1980/1981–1994/1994**	**1995/1996–2008/2009**	**−0.309**	**0.760**	**2.052**	**No**	**92**	**−0.546**	**0.585**	**59**	**No**
	1980/1981–1989/1990	**1990/1991–1999/2000**	**0.246**	**0.809**	**2.101**	**No**	**48**	**0.113**	**0.910**	**23**	**No**
	1980/1981–1989/1990	**2000/2001–2008/2009**	**−0.138**	**0.892**	**2.110**	**No**	**42**	**−0.204**	**0.838**	**20**	**No**
	1990/1991–1999/2000	**2000/2001–2008/2009**	**−0.421**	**0.679**	**2.110**	**No**	**39**	**−0.449**	**0.653**	**20**	**No**

Dev	1980/1981–1994/1994	1995/1996–2008/2009	−2.476	0.020	2.052	Yes	58	−2.029	0.042	59	Yes
	1980/1981–1989/1990	**1990/1991–1999/2000**	**−1.309**	**0.207**	**2.101**	**No**	**37**	**−0.945**	**0.345**	**23**	**No**
	1980/1981–1989/1990	**2000/2001–2008/2009**	**−1.375**	**0.187**	**2.110**	**No**	**31**	**−1.102**	**0.270**	**20**	**No**
	1990/1991–1999/2000	**2000/2001–2008/2009**	**−0.112**	**0.912**	**2.110**	**No**	**43**	**−0.122**	**0.903**	**20**	**No**
Pnes	**1980/1981–1994/1994**	**1995/1996–2008/2009**	**0.340**	**0.736**	**2.052**	**No**	**105**	**0.022**	**0.983**	**59**	**No**
	1980/1981–1989/1990	1990/1991–1999/2000	2.564	0.020	2.101	Yes	20	2.230	0.026	23	Yes
	1980/1981–1989/1990	**2000/2001–2008/2009**	**0.847**	**0.409**	**2.110**	**No**	**36**	**0.694**	**0.488**	**20**	**No**
	1990/1991–1999/2000	2000/2001–2008/2009	−2.301	0.034	2.110	Yes	19	−2.082	0.037	20	Yes
Dnes	**1980/1981–1994/1994**	**1995/1996–2008/2009**	**−1.977**	**0.058**	**2.052**	**No**	**65**	**−1.724**	**0.085**	**59**	**No**
	1980/1981–1989/1990	**1990/1991–1999/2000**	**−0.071**	**0.944**	**2.101**	**No**	**46**	**0.265**	**0.791**	**23**	**No**
	1980/1981–1989/1990	**2000/2001–2008/2009**	**−1.092**	**0.290**	**2.110**	**No**	**31**	**−1.102**	**0.270**	**20**	**No**
	1990/1991–1999/2000	**2000/2001–2008/2009**	**−0.999**	**0.332**	**2.110**	**No**	**34**	**−0.857**	**0.391**	**20**	**No**
Ppin	**1980/1981–1994/1994**	**1995/1996–2008/2009**	**0.219**	**0.828**	**2.052**	**No**	**101**	**0.153**	**0.879**	**59**	**No**
	1980/1981–1989/1990	**1990/1991–1999/2000**	**0.109**	**0.915**	**2.101**	**No**	**48**	**0.113**	**0.910**	**23**	**No**
	1980/1981–1989/1990	**2000/2001–2008/2009**	**0.308**	**0.762**	**2.110**	**No**	**42**	**0.204**	**0.838**	**20**	**No**
	1990/1991–1999/2000	**2000/2001–2008/2009**	**0.225**	**0.825**	**2.110**	**No**	**42**	**0.204**	**0.838**	**20**	**No**

(continued)

Table 9 (continued)

x	Subseries 1	Subseries 2	*t*-test				Mann–Whitney *U* test				
			t	$P(T<=t)$ two tail	t-critical two tail	Significance	U	z	p-value	U-critical	Significance
Dpin	**1980/1981–1994/1994**	**1995/1996–2008/2009**	**−0.479**	**0.636**	**2.052**	**No**	**93**	**−0.502**	**0.616**	**59**	**No**
	1980/1981–1989/1990	**1990/1991–1999/2000**	**−0.543**	**0.594**	**2.101**	**No**	**43**	**−0.491**	**0.623**	**23**	**No**
	1980/1981–1989/1990	**2000/2001–2008/2009**	**−0.298**	**0.769**	**2.110**	**No**	**41**	**−0.286**	**0.775**	**20**	**No**
	1990/1991–1999/2000	**2000/2001–2008/2009**	**0.173**	**0.864**	**2.110**	**No**	**44**	**0.041**	**0.967**	**20**	**No**
Pstr	**1980/1981–1994/1994**	**1995/1996–2008/2009**	**−1.781**	**0.086**	**2.052**	**No**	**61**	**−1.898**	**0.058**	**59**	**No**
	1980/1981–1989/1990	**1990/1991–1999/2000**	**1.086**	**0.292**	**2.101**	**No**	**39**	**0.794**	**0.427**	**23**	**No**
	1980/1981–1989/1990	**2000/2001–2008/2009**	**−1.780**	**0.093**	**2.110**	**No**	**22**	**−1.837**	**0.066**	**20**	**No**
	1990/1991–1999/2000	2000/2001–2008/2009	−3.115	0.006	2.110	Yes	15	−2.409	0.016	20	Yes
Dstr	**1980/1981–1994/1994**	**1995/1996–2008/2009**	**−1.424**	**0.166**	**2.052**	**No**	**75**	**−1.287**	**0.198**	**59**	**No**
	1980/1981–1989/1990	**1990/1991–1999/2000**	**0.149**	**0.883**	**2.101**	**No**	**48**	**0.113**	**0.910**	**23**	**No**
	1980/1981–1989/1990	**2000/2001–2008/2009**	**−0.552**	**0.588**	**2.110**	**No**	**38**	**−0.531**	**0.596**	**20**	**No**
	1990/1991–1999/2000	**2000/2001–2008/2009**	**−0.645**	**0.527**	**2.110**	**No**	**41**	**−0.286**	**0.775**	**20**	**No**

The results in bold indicate which series are stationary (5% significance level)

Table 10 Test statistics of the trend tests for the annual discharge and precipitation

Test	Mann–Kendall		Spearman's Rho	
	Z	Significance (5% confidence level)	D	Significance (5% confidence level)
Pach	−1.107	No	−1.077	No
Dach	0.469	No	0.571	No
Pal	0.394	No	0.344	No
Dal	0.544	No	0.683	No
Pax	−0.206	No	−0.339	No
Dax	0.394	No	0.315	No
Pev	0.000	No	−0.013	No
Dev	1.182	No	1.243	No
Pnes	−0.544	No	−0.725	No
Dnes	0.825	No	1.035	No
Ppin	−0.732	No	−0.769	No
Dpin	0.281	No	0.255	No
Pstr	1.182	No	1.361	No
Dstr	0.431	No	0.516	No

Critical value at 5% confidence level: 1.96

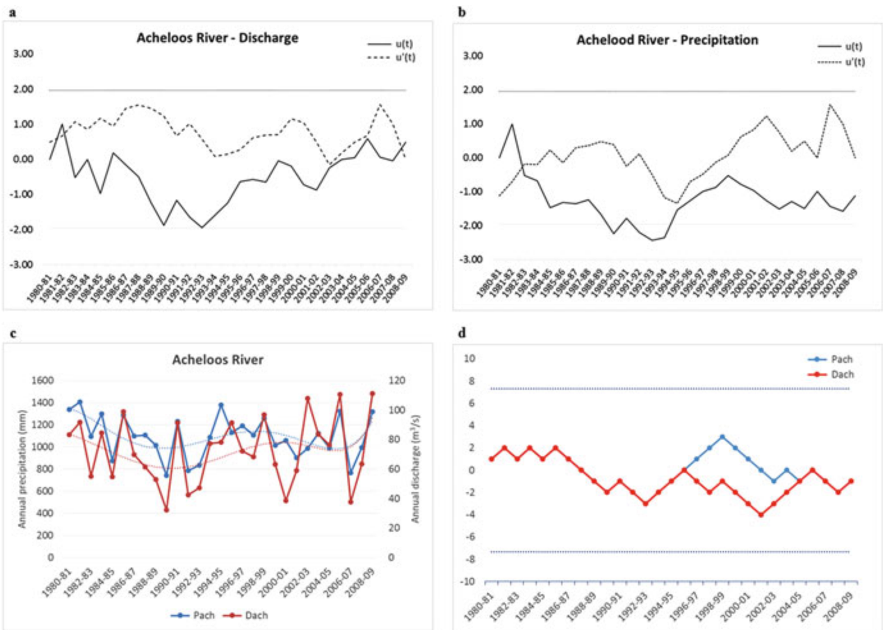


Fig. 6 SMK test for annual discharge (**a**), annual precipitation (**b**) and variation of annual precipitation and discharge (**c**) and CUSUM test for discharge and precipitation (**d**) of Acheloos River

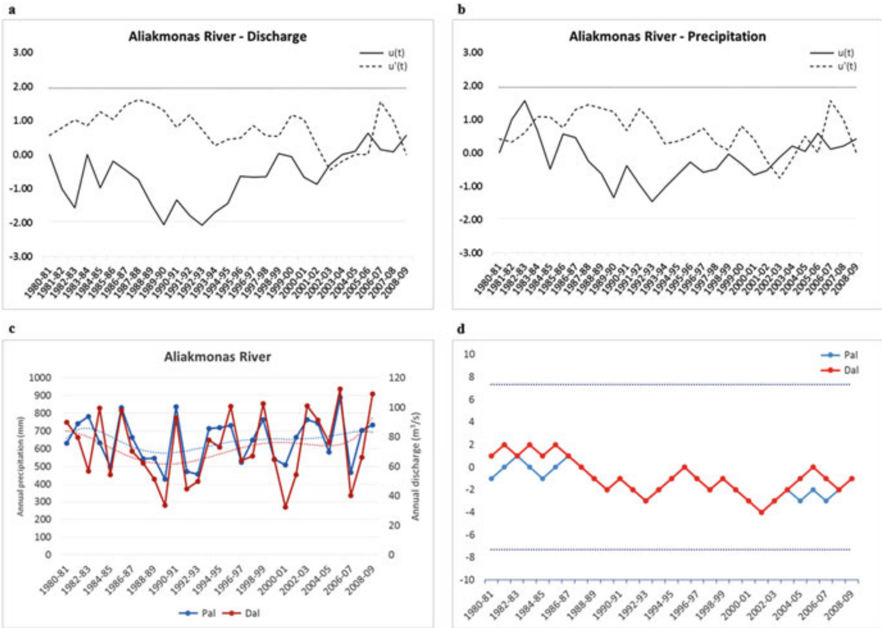


Fig. 7 SMK test for annual discharge (**a**), annual precipitation (**b**) and variation of annual precipitation and discharge (**c**) and CUSUM test for discharge and precipitation (**d**) of Aliakmonas River

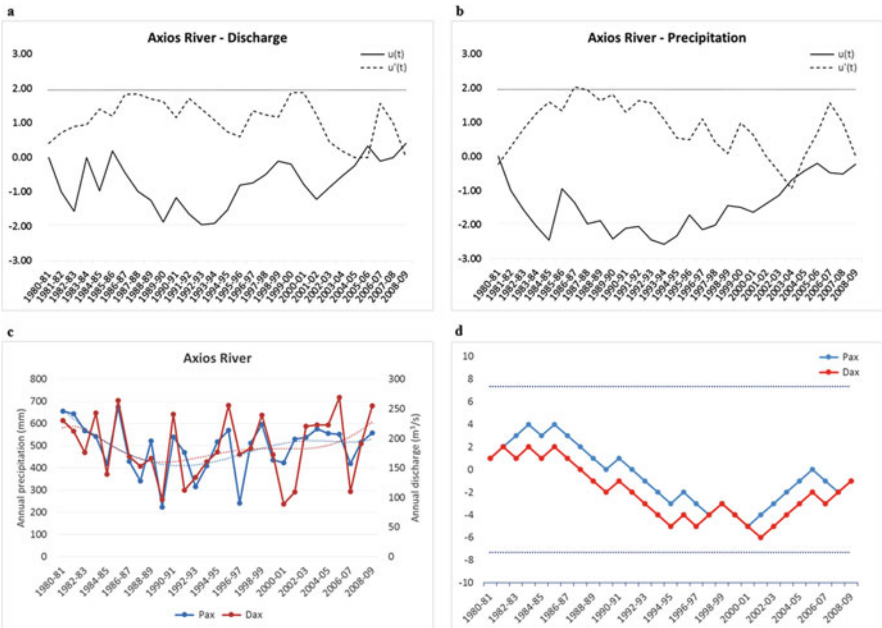


Fig. 8 SMK test for annual discharge (**a**), annual precipitation (**b**) and variation of annual precipitation and discharge (**c**) and CUSUM test for discharge and precipitation (**d**) of Axios River

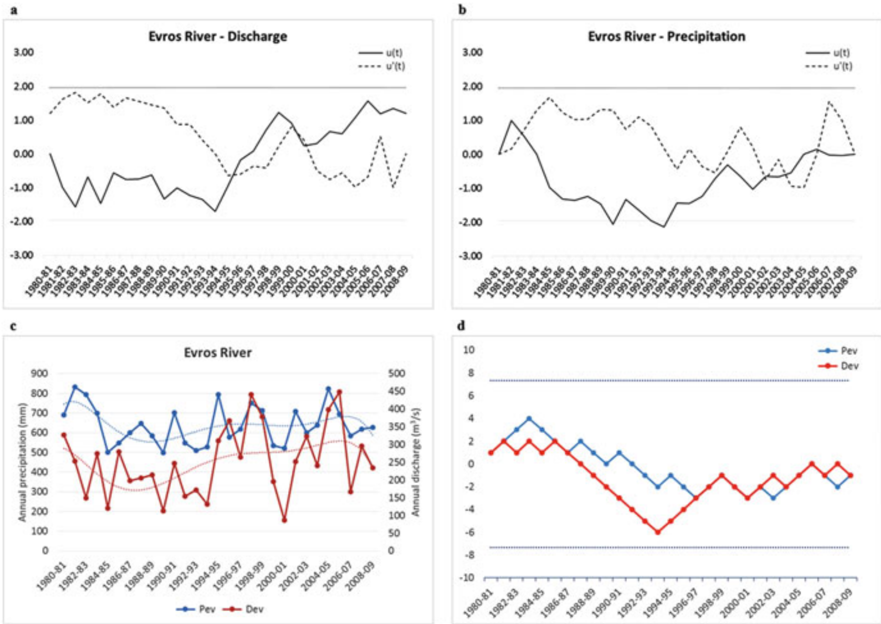


Fig. 9 SMK test for annual discharge (**a**), annual precipitation (**b**) and variation of annual precipitation and discharge (**c**) and CUSUM test for discharge and precipitation (**d**) of Evros River

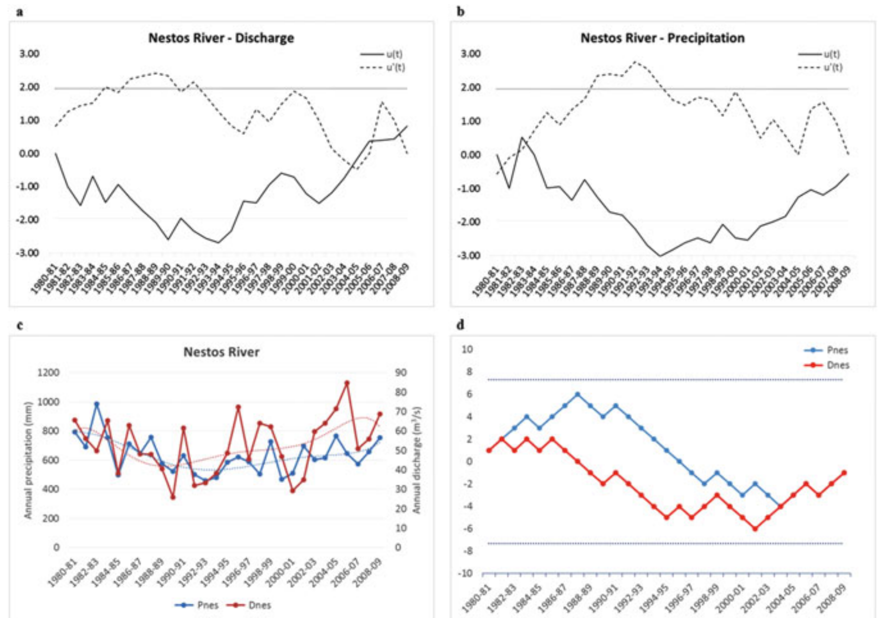


Fig. 10 SMK test for annual discharge (**a**), annual precipitation (**b**) and variation of annual precipitation and discharge (**c**) and CUSUM test for discharge and precipitation (**d**) of Nestos River

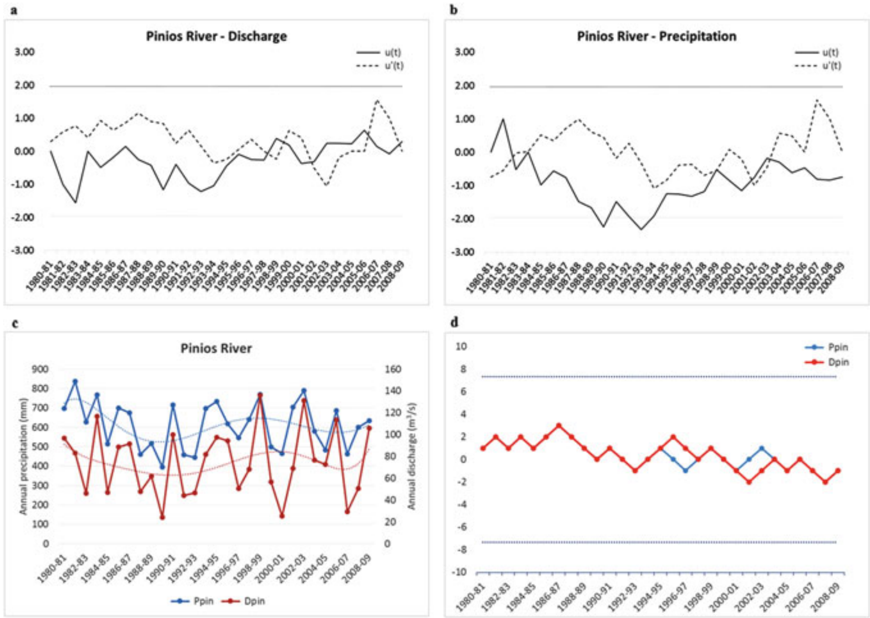


Fig. 11 SMK test for annual discharge (**a**), annual precipitation (**b**) and variation of annual precipitation and discharge (**c**) and CUSUM test for discharge and precipitation (**d**) of Pinios River

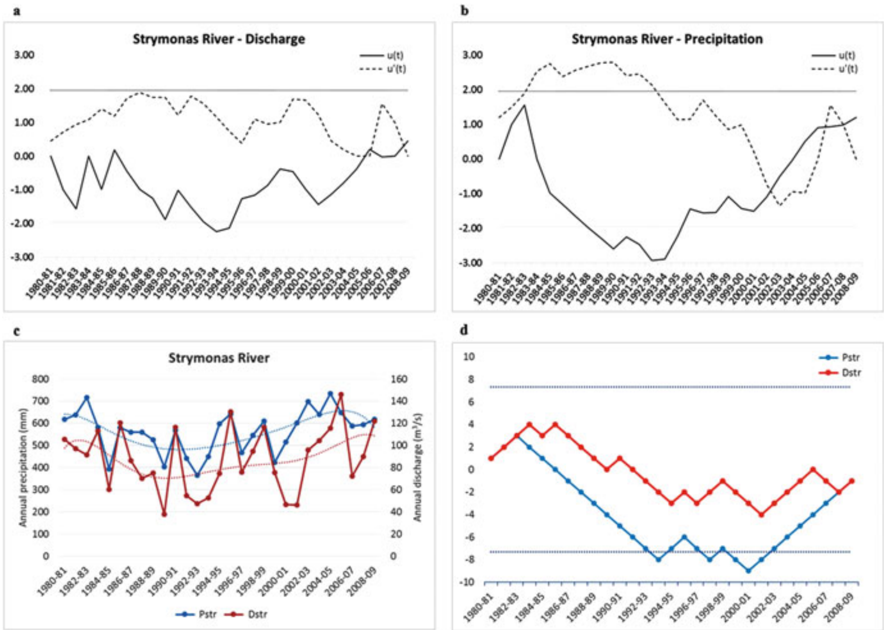


Fig. 12 SMK test for annual discharge (**a**), annual precipitation (**b**) and variation of annual precipitation and discharge (**c**) and CUSUM test for discharge and precipitation (**d**) of Strymonas River

Table 11 Results of change point analysis and Sen's slope estimation

River	Precipitation						Discharge					
	Period		Trend	Sen's slope			Period		Trend	Sen's slope		
				mm/year	km^3/year	Sign				m^3/s/year	km^3/year	Sign
Acheloos	1981/1982	1992/1993	Decrease	−49.234	−0.318	*	1981/1982	1992/1993	Decrease	−3.675	−0.116	**
	1992/1993	1998/1999	Increase	35.160	0.227		1992/1993	1998/1999	Increase	4.230	0.133	
	1998/1999	2002/2003	Decrease	−61.997	−0.401		1998/1999	2001/2002	Decrease	−18.640	−0.588	
	2002/2003	2008/2009	Increase	1.749	0.011		2001/2002	2008/2009	Increase	0.662	0.021	
Aliakmonas	1981/1982	1992/1993	Decrease	−23.905	−0.154		1981/1982	1992/1993	Decrease	−2.566	−0.081	**
	1992/1993	1998/1999	Increase	11.128	0.072		1992/1993	1998/1999	Increase	3.508	0.111	
	1998/1999	2000/2001	Decrease	−127.456	−0.824		1998/1999	2000/2001	Decrease	−34.969	−1.103	
	2000/2001	2008/2009	Increase	14.199	0.092		2000/2001	2008/2009	Increase	4.288	0.135	
Axios	1981/1982	1993/1994	Decrease	−21.191	−0.137	*	1981/1982	1992/1993	Decrease	−7.110	−0.224	**
	1993/1994	1998/1999	Increase	25.755	0.166		1992/1993	1998/1999	Increase	15.654	0.494	**
	1998/1999	2000/2001	Decrease	−85.865	−0.555		1998/1999	2001/2002	Decrease	−55.386	−1.747	
	2000/2001	2008/2009	Increase	3.596	0.023		2001/2002	2008/2009	Increase	6.164	0.194	
Evros	1981/1982	1993/1994	Decrease	−18.737	−0.121	*	1981/1982	1993/1994	Decrease	−7.022	−0.221	
	1993/1994	1998/1999	Increase	37.081	0.240		1993/1994	1998/1999	Increase	43.991	1.387	
	1998/1999	2000/2001	Decrease	−95.822	−0.619		1998/1999	2000/2001	Decrease	−145.933	−4.602	
	2000/2001	2008/2009	Increase	4.157	0.027		2000/2001	2008/2009	Increase	13.580	0.428	
Nestos	1981/1982	1993/1994	Decrease	−29.080	−0.188	***	1981/1982	1993/1994	Decrease	−1.740	−0.055	*
	1993/1994	1998/1999	Increase	35.294	0.228		1993/1994	1998/1999	Increase	4.797	0.151	
	1998/1999	2000/2001	Decrease	−108.228	−0.699		1998/1999	2000/2001	Decrease	−16.457	−0.519	
	2000/2001	2008/2009	Increase	12.876	0.083		2000/2001	2008/2009	Increase	4.887	0.154	
Pinios	1981/1982	1992/1993	Decrease	−23.905	−0.154		1981/1982	1992/1993	Decrease	−2.566	−0.081	**
	1992/1993	1998/1999	Increase	11.128	0.072		1992/1993	1998/1999	Increase	3.508	0.111	
	1998/1999	2000/2001	Decrease	−127.456	−0.824		1998/1999	2000/2001	Decrease	−34.969	−1.103	
	2000/2001	2008/2009	Increase	14.199	0.092		2000/2001	2008/2009	Increase	4.288	0.135	
Strymonas	1981/1982	1992/1993	Decrease	−20.493	−0.132	***	1981/1982	1993/1994	Decrease	−4.411	−0.139	*
	1992/1993	1998/1999	Increase	35.773	0.231		1993/1994	1998/1999	Increase	10.488	0.331	
	1998/1999	2000/2001	Decrease	−47.030	−0.304		1998/1999	2001/2002	Decrease	−26.122	−0.824	**
	2000/2001	2008/2009	Increase	3.039	0.020		2001/2002	2008/2009	Increase	7.887	0.249	

*Statistically significant at 5% confidence level
**Statistically significant at 10% confidence level
***Statistically significant at 1% confidence level

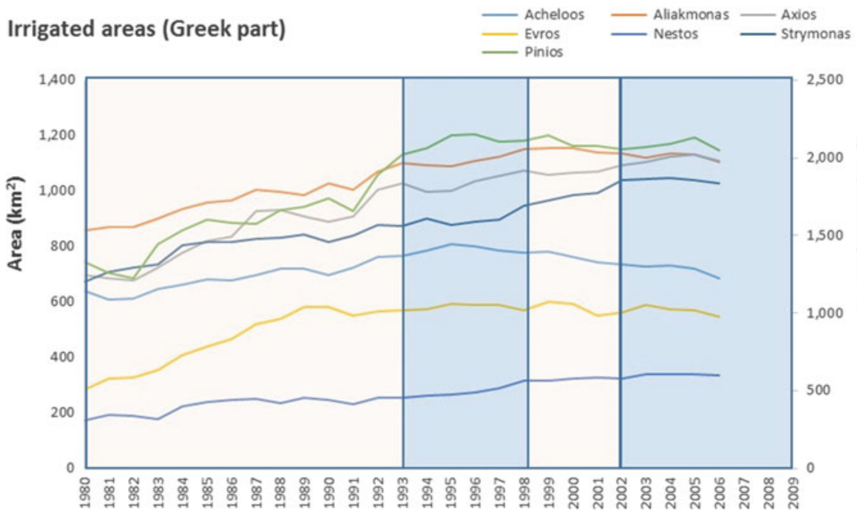


Fig. 13 Irrigated areas in the river catchments under study (only the Greek part for the transboundary rivers)

In all cases, the trend analysis of annual discharge and precipitation indicated similar characteristics and two decreasing periods can be identified (Figs. 6, 7, 8, 9, 10, 11 and 12; Table 11). Based on the statistical tests applied, there is a decrease of the annual discharge for the period 1981/1982–1992/1993 or 1993/1994, which is followed by an increasing trend until 1998/1999. A second decreasing period is then identified for the period 1998/1999 until 2000/2001–2002/2003. Finally, an increasing period is identified until 2008/2009. These fluctuations coincide with extended wet and dry periods that have been clearly recorded in the meteorological history of the country, such as the persistent drought of 1989–1993 and the drought of 2003 [49], with severe socio-economic impacts (e.g. 1988–1993). At that period, special policy measures were implemented to impose water-saving practices and prevent water overconsumption (e.g. prohibit individual car washing, introduction of a stepwise billing system, etc.). An extensive drought of a lesser degree followed in the beginning of the century, which also caused an increase in the domestic water prices and a decrease in the agricultural productivity, while after 2003 there is a prolonged wet period which allowed for easing the rates of water consumption.

The aforementioned trends are clearly visible in all river catchments which means that these wet/dry periods were large-scale events (national level or even beyond that) but the response rates from river to river differed significantly. Nestos and Strymonas rivers presented the most decreasing trends (in both discharge and precipitation) in the late 1980s, while Evros and Aliakmonas demonstrated the most increasing trends during the last decade.

Moreover, the fluctuations and trends in discharge follow, to a great extent, those of precipitation, and this is also verified by the CUSUM diagrams, which usually coincide during most of the study period for the two variables.

Finally, Sen's slope (Table 11) was employed in order to estimate the change per unit time (km^3/year) of the trends observed in discharge and precipitation time series of all rivers. A negative slope indicates a decrease of the annual precipitation of discharge for the period concerned and therefore a downward trend, while a positive slope indicates an increase and an upward trend. Based on the results, in the case of Axios, Evros and Strymonas rivers, the discharge decreasing trend is more pronounced compared to the precipitation decreasing trend, for the period 1981/1982–1993/1994. Similarly, in all cases except Nestos River, the decrease of the discharge is more pronounced compared to the precipitation decreasing trend, during the period 1998/1999–2000/2001 to 2002/2003. These findings could be attributed to increasing irrigation demands during these periods to the specific watersheds that in combination with limited precipitation rate intensified the decreasing trend of river discharge (Fig. 13).

5 Conclusions

The lack of appropriate discharge measurements at country level imposes many difficulties to identify alarming hydrologic trends and undertake timely mitigation and restoration measures. For this reason, the few systematically recorded but fragmented past hydrologic data of the main Greek rivers have been used to assess the validity of a regional hydrological model in order to use its long-term output (1980–2009) for analysing the dominant hydrologic trends. It is for this purpose that the discharge data from the E-HYPE pan-European hydrological model have been used in combination with precipitation data from the Hellenic National Meteorological Service; the Ministry of Environment, Energy and Climate Change of Greece; the Republic Hydrometeorological Institute of FYROM; the National Institute of Meteorology and Hydrology of Bulgaria and the Turkish State Meteorological Service. The main purpose of this study was to identify the dominant trends in the discharge and precipitation time series over the last 3 decades as well as to study the interrelationship between these two parameters.

The main output is that there is no consistent, single trend for the entire study period for any of the investigated rivers while there are some wet and dry periods in the data which are very clear in all catchments and coincide at a temporal level. The main dry periods were at the end of the 1980s and the beginning of 2000s. There is also a prolonged wet period during the last decade for all study catchments while all the hydrometeorologic time series are stationary apart from Evros, Nestos and Strymonas rivers.

The relationship between precipitation and discharge is quite strong in Aliakmonas and Pinios rivers and relatively weak in Evros and Nestos rivers. Nevertheless, precipitation and discharge present very similar fluctuation patterns throughout the study period as the CUSUM diagrams indicate, while all the time series were homogeneous apart from Evros River annual discharge and Nestos River annual precipitation.

A more in-depth analysis is required to understand, conceptualise and record the factors that contribute to discharge apart from precipitation (e.g. irrigation, domestic use, infiltration, dam operation, etc.). In order for this study to be reliable and useful, real discharge measurements should be taken prior to the analysis by establishing a (transboundary where relevant) monitoring network that can act with a double purpose: to provide an early warning system for natural–man-made disasters such as floods and droughts and to offer the necessary, scientific information for the development and implementation of integrated water resources management plans.

Annexes

Annex I

Statistical criteria of E-HYPE model performance by comparing the observed (o) and predicted (p) data of a sample size (n)

Mean error ME: $\mathrm{ME} = \frac{\sum_{i=1}^{n}(o_i - p_i)}{n}$

Mean absolute error MAE: $\mathrm{ME} = \frac{\sum_{i=1}^{n}|o_i - p_i|}{n}$

Mean absolute percentage error MAPE: $\mathrm{ME} = \frac{\sum_{i=1}^{n}\left|\frac{o_i - p_i}{o_i}\right|}{n} \times 100$

Root mean squared error RMSE: $\mathrm{ME} = \sqrt{\frac{\sum_{i=1}^{n}(p_i - o_i)^2}{n}}$

Pearson's correlation coefficient R: $R = \frac{\sum_{i=1}^{n}(p_i - \overline{p})(o_i - \overline{o})}{\sqrt{\sum_{i=1}^{n}(p_i - \overline{p})^2}\sqrt{\sum_{i=1}^{n}(o_i - \overline{o})^2}}$

Squared correlation coefficient R^2: $R^2 = R^2$

Nash–Sutcliffe coefficient of efficiency Nr: $\mathrm{Nr} = 1 - \frac{\sum_{i=1}^{n}(o_i - p_i)^2}{\sum_{i=1}^{n}(o_i - \overline{o})^2}$

Annex II

Normality Tests

Kolmogorov–Smirnov (KS) test: $\mathrm{KS} = \sup_x|\mathrm{F}^*(x) - \mathrm{Fn}(x)|$, where sup stands for supremum, $\mathrm{Fn}(x)$ is theoretical cumulative distribution function of the normal distribution function and $F^*(x)$ is the normal empirical distribution function of the data, with known mean μ and standard deviation σ.

Lilliefors (LF) test: $\mathrm{LF} = \max_x|F^*(x) - Sn(x)|$, where $Sn(x)$ is the sample cumulative distribution function of the normal distribution function and $F^*(x)$ is the empirical distribution function, with the sample mean $\mu = \bar{x}$ and the sample variance s^2 defined with denominator $n - 1$.

Shapiro–Wilk (SW) test: $\mathrm{SW} = \frac{\left(\sum_{i=1}^{n} a_i x_i\right)^2}{\sum_{i=1}^{n} (x_i - \bar{x})^2}$, where x_i stands for ordered (increasing ordered) sample values and a_i stands for constants generated from the means, variances and covariances of the order statistics of a sample of size n from a normal distribution.

Homogeneity Tests

von Neumann test: $N = \frac{\sum_{i=1}^{n-1} (x_i - x_{i+1})^2}{\sum_{i=1}^{n} (x_i - \bar{x})^2}$, where x_i is the hydrologic variable constituting the sequence in time, n is the total number of hydrologic records and $\bar{x}$ is the average of x_i.

Cumulative deviations test: Sensitivity to the departures from homogeneity is defined by the following statistic:

$Q = \max_{0 \le k \le n} \left|S_k^{**}\right|$, where S_k^{**} is the rescaled adjusted partial sums.

$S_k^{**} = S_k^*/D_x$, $k = 1, 2, \ldots, n$, where $S_k^* = \Sigma_{i=1}^{k} (x_i - \bar{x})$, $k = 1, 2, \ldots, n$, and D_x the sample standard deviation.

High values of Q are an indication for non-homogeneity.

The homogeneity can also be tested with the following statistic:

$$R = \max_{0 \le k \le n} \left|S_k^{**}\right| - \min_{0 \le k \le n} \left|S_k^{**}\right|$$

Bayesian Test: $U = \frac{1}{n(n+1)} \Sigma_{k=1}^{n-1} \left(S_k^{**}\right)^2$, for p_k independent of k.

$A = \Sigma_{i=1}^{n-1} \left(Z_k^{**}\right)^2$, $k = 1, 2, \ldots, n$, for p_k proportional to $[k(n-k)]^{-1}$. Z_k^{**} is the weighted rescaled partial sums, $Z_k^{**} = \left[\{k(n-k)\}^{-1/2} S_k^*\right] / D_x$.

Stationarity Tests

t-test: To apply this test, the annual time series is divided into two (or more) subseries of size n_1 and n_2 ($n_1 + n_2 = n$):

$t_s = \frac{|\bar{x}_2 - \bar{x}_1|}{S\sqrt{\frac{1}{n_1} + \frac{1}{n_2}}}$, $S = \sqrt{\frac{(n_1 - 1)s_1^2 + (n_2 - 1)s_2^2}{n - 2}}$, where $\bar{x}_1$, $\bar{x}_2$, s_1^2 and s_2^2 are the estimated means and variances of the first and the second subseries, respectively.

Mann–Whitney test: To apply this test, the annual time series n_t is divided into two (or more) subseries of size n_1 and n_2 ($n_1+n_2=n$), and a new series z_t ($t=1, 2, \ldots, n$) is defined by arranging the original data (n_t) in increasing order of magnitude:

$u = \frac{\Sigma_{t=1}^{n_1} R(n_t) - n_1(n_1+n_2+1)/2}{[n_1 n_2 (n_1+n_2+1)/12]^{1/2}}$, where $R(n_t)$ is the rank of the observation n_t in ordered series z_t.

Trend

Mann–Kendall test: The Mann–Kendall statistic S compares each value of the series (x_t) with all subsequent values (x_{t+1}) and is defined as

$S = \Sigma_{t'=1}^{n-1} \Sigma_{t=t'+1}^{n} \operatorname{sgn}\left(xt - x_{t'}\right)$, where sgn is the signum function,

$$\operatorname{sgn}\left(xt - x_{t'}\right) = \begin{cases} 1, \text{ if } x_t > x_{t'} \\ 0, \text{ if } x_t = x_{t'} \\ -1, \text{ if } x_t < x_{t'} \end{cases}$$

Based on Mann [41] and Kendall [40], when $n \geq 8$, the statistic S is approximately normally distributed with the mean m and the variance V as follows: $E(S) = 0$, $V(S) = \frac{1}{18}\left[n(n-1)(2n+5) - \Sigma_{i=1}^{g} e_i(e_i-1)(2e_i+5)\right]$, g is the number of tied groups, and e_i is the number of data in the ith tied group.

The standardised test statistic Z is defined as $Z = \frac{S+m}{\sqrt{V(S)}}$.

Spearman's Rho: The Spearman's Rho D statistic is defined as

$D = 1 - \frac{6\Sigma_{1=1}^{n}[R(X_i)-i]^2}{n(n^2-1)}$, where $R(X_i)$ is the rank of ith observation X_i in the sample size n.

Under the null hypothesis that the time series has no trend, it can be shown that the statistic t_s has a Student's t-distribution with n–2 degrees of freedom. Here, t_s is defined as $t_s = D\sqrt{\frac{n-2}{1-D^2}}$.

Sequential Version of the Mann–Kendall Test (Mann–Kendall Rank Correlation Test): The sequential version of the Mann–Kendall test is calculated so that rank (x_i) > rank (x_j) ($i > j$). The t statistic is calculated as $t = \Sigma_{i=1}^{n} n_i$. The distribution of t is assumed to be asymptotically normal with the following expectation: $E(t) = \mu = \frac{n(n-1)}{4}$ and $\operatorname{Var}(t) = \sigma^2 = \frac{n(n-1)(2n+5)}{72}$.

The null hypothesis that there is no trend is rejected for high values of the reduced variable $|u(t)|$, which is calculated as $u(t) = \frac{t-E(t)}{\sqrt{\operatorname{Var}(t)}}$. The statistic $u'(t)$ is computed backwards starting from the end of the time series.

CUSUM Test: The test statistic Vk is defined as $V_k = \Sigma_{i=1}^{k} \operatorname{sgn}(x_i - x_{\text{median}})$, $k = 1,2, \ldots, n$, where x_{median} is the median value of the x_i data set and sgn(x).

Sen's Slope Estimator: The Sen's slope estimation test is defined for a season g as $\beta = \text{Median}\left(\frac{x_i - x_j}{i - j}\right)$, $i < j$, where Q is the slope between points x_i and x_j, x_i is data measurement at time i and x_j is data measurement at time j.

It is defined as the estimator β which is the median overall combination of record pairs for the whole data set and is resistant/robust to the extreme observations or outliers. The positive value of the β connotes the slope of the upward trend and negative value for the downward trend [29].

References

1. Iglesias A, Garrote L, Flores F, Moneo M (2007) Challenges to manage the risk of water scarcity and climate change in the Mediterranean. Water Resour Manag 21:775–788
2. Zacharias I, Dimitriou E, Koussouris T (2005) Integrated water management scenarios for wetland protection: application in Trichonis Lake. Environ Model Softw 20:177–185
3. Sofios S, Arabatzis G, Baltas E (2008) Policy for management of water resources in Greece. Environmentalist 28:185–194
4. Moschou EC, Batelis SC, Dimakos Y, et al (2013) Spatial and temporal rainfall variability over Greece. In: 5th EGU Leonardo conference – Hydrofractals 2013 – STAHY'13, Kos Island, Greece, European Geosciences Union, International Association of Hydrological Sciences, International Union of Geodesy and Geophysics
5. Hellenic Ministry for the Environment Physical Planning and Public Works (2008) National reporting to the fourteenth & fifteenth sessions of the Commission for Sustainable Development of the United Nations (UNCSD 14-UNCSD 15), Athens
6. Mimikou M, Baltas E (2013) Assessment of climate change impacts in Greece: a general overview. Am J Clim Chang 2:46–56
7. Atay I (2012) Water resources management in Greece: perceptions about water problems in the Nafplion area. Independent thesis Advanced level, Stockholm University
8. Jarboe JE, Haan CT (1974) Calibrating a water yield model for small ungaged watersheds. Water Resour Res 10:256–262
9. Machiwal D, Jha MK (2006) Time series analysis of hydrologic data for water resources planning and management: a review. J Hydrol Hydromech 54:237–257
10. Burn DH, Hag Elnur MA (2002) Detection of hydrologic trends and variability. J Hydrol 255:107–122
11. Liarikos K, Maragou P, Papagiannis TH (2012) Nationwide land cover mapping and land cover changes in Greece during 1987–2007. WWF, Athens
12. European Environmental Agency (EEA) (2014) Corine Land Cover 2006. http://www.eea.europa.eu. Accessed 10 Apr 2014
13. Skoulikidis N, Economou A, Gritzalis K, Zogaris S (2009) Rivers of the Balkans. In: Tockner K, Uehlinger U, Robinson CT (eds) Rivers of Europe, 1st edn. Elsevier, London
14. Donevska K (2006) Report on second communication on climate and climate changes and adaptation in the Republic of Macedonia-Section: vulnerability assessment and adaptation for water resources sector. Ministry of Environment and Physical Planning, Skopje
15. Koutsoyiannis D, Andreadakis A, Mavrodimou R et al (2008) Support on the compilation of the national programme for water resources management and preservation. National Technical University of Athens-Department of Water Resources and Environmental Engineering, Athens
16. National Statistical Institute of the Republic of Bulgaria (NSI) (2011) Environment 2009. National Statistical Institute of the Republic of Bulgaria, Sofia. http://www.nsi.bg/sites/default/files/files/publications/Environment2009.pdf. Accessed 3 May 2014

17. Yildiz D (2008) Water report- our need for a national water policy. National Association of Industrialists and Businessmen-USİAD, Ertem Matbaa, Ankara
18. Climate Prediction Center/National Centers for Environmental Prediction/National Weather Service/NOAA/U.S. Department of Commerce (2014) CPC global summary of day/month observations, 1979-continuing. Research Data Archive at the National Center for Atmospheric Research, Computational and Information Systems Laboratory. http://rda.ucar.edu/datasets/ds512.0/. Accessed 16 Apr 2014
19. Arheimer B, Dahné J, Donnelly C et al (2012) Water and nutrient simulations using the HYPE model for Sweden vs. the Baltic Sea basin – influence of input-data quality and scale. Hydrol Res 43:315–329
20. Strömqvist J, Arheimer B, Dahné J et al (2012) Water and nutrient predictions in ungauged basins: set-up and evaluation of a model at the national scale. Hydrol Sci J 57:229–247
21. Beven K (2012) Rainfall-runoff modelling: the primer. Wiley, Chichester
22. Krause P, Boyle DP, Bäse F (2005) Comparison of different efficiency criteria for hydrological model assessment. Adv Geosci 5:89–97
23. Singh J, Knapp HV, Demissie M (2004) Hydrologic modeling of the Iroquois River watershed using HSPF and SWAT. Illinois State Water Survey, Champaign
24. Nash JE, Sutcliffe JV (1970) River flow forecasting through conceptual models. Part I – a discussion of principles. J Hydrol 10:282–290
25. Legates DR, McCabe GJ Jr (1999) Evaluating the use of "goodness-of-fit" measures in hydrologic and hydroclimatic model validation. Water Resour Res 35:233–241
26. Kolmogorov AN (1933) Sulla determinazione empirica di una legge di distribuzione. G dell'Istituto Ital degli Attuari 4:83–91
27. Lilliefors HW (1967) On the Kolmogorov–Smirnov test for normality with mean and variance unknown. J Am Stat Assoc 62:399–402
28. Shapiro ASS, Wilk MB (1965) An analysis of variance test for normality (complete samples). Biometrika 52:591–611
29. Machiwal D, Jha M (2012) Hydrologic time series analysis: theory and practice. Springer, The Netherlands
30. Massey FJ Jr (1951) The Kolmogorov–Smirnov test for goodness of fit. J Am Stat Assoc 46:68–78
31. Fernando DAK, Jayawardena AW (1994) Generation and forecasting of monsoon rainfall data. In: 20th WEDC conference on affordable water supply and sanitation, Colombo
32. Von Neumann J (1941) Distribution of the ratio of the mean square successive difference to the variance. Ann Math Stat 12:367–395
33. Buishand TA (1982) Some methods for testing the homogeneity of rainfall records. J Hydrol 58:11–27
34. Chernoff H, Zacks S (1964) Estimating the current means of a normal distribution which is subjected to change in time. Ann Math Stat 35:999–1018
35. Gardner LA Jr (1969) On detecting changes in the mean of normal variates. Ann Math Stat 40:116–126
36. Dahmen ER, Hall MJ (1990) Screening of hydrological data: tests for stationarity and relative consistency. ILRI publication no. 49. ILRI, Wageningen
37. Snedecor GW, Cochran WG (1989) Statistical methods. Wiley-Blackwell, New Jersey
38. Haan CT (2002) Statistical methods in hydrology. Iowa State University Press, Ames
39. Shahin M, Van Oorschot HJL, De Lange SJ (1993) Statistical analysis in water resources engineering. A.A. Balkema, Rotterdam
40. Kendall MG (1975) Rank correlation methods. Charles Griffin and Company, London
41. Mann HB (1945) Nonparametric tests against trend. Econometrica 13:245–259
42. Lehmann EL (1975) Nonparametrics: statistical methods based on ranks. Holden-Day, San Francisco
43. Sneyers R (1990) On the statistical analysis of series of observations, Technical note no. 143. World Meteorological Organization, Geneva

44. Spearman C (1904) The proof and measurement of association between two things. Am J Psychol 15:72–101
45. Yue S, Pilon P, Cavadias G (2002) Power of the Mann–Kendall and Spearman's rho tests for detecting monotonic trends in hydrological series. J Hydrol 259:254–271
46. McGilchrist CA, Woodyer KD (1975) Note on a distribution-free CUSUM technique. Technometrics 17:321–325
47. Sen PK (1968) Estimates of the regression coefficient based on Kendall's Tau. J Am Stat Assoc 63:1379–1389
48. Moriasi DN, Arnold JG, Van Liew MW et al (2007) Model evaluation guidelines for systematic quantification of accuracy in watershed simulations. Trans ASABE 50:885–900
49. Vangelis H, Spiliotis M, Tsakiris G (2010) Drought severity assessment based on bivariate probability analysis. Water Resour Manag 25:357–371

Agro-Industrial Wastewater Pollution in Greek River Ecosystems

Ioannis Karaouzas

Abstract In this chapter, the characteristics and environmental impacts of wastewaters from the major agricultural industries on the river ecosystems of Greece are reviewed and discussed, focusing especially on olive mills, orange juice processing factories and cheese processing factories. The high organic load, suspended solids and nutrients of these wastewaters, as well as their toxicity, have deteriorated river water quality and the ecological status of many running waters of Greece. Among the most common effects are eutrophication, the decline of fish and invertebrate populations, species richness loss and the consequent reduction of the river capacity for moderating the effects of polluting substances through internal mechanisms of self-purification. The organic load of the wastewaters, substrate contamination (sewage bacteria) and distance from the wastewater discharge outlet appear to be the most important factors affecting macroinvertebrate assemblages, while typology (i.e. slope, altitude), hydrology (i.e. permanent, intermittent), intensity and volume of the wastewater are the most important determinants of self-purification processes. As these industries are usually located near small-sized streams that are not significantly considered in the Water Framework Directive 2000/60/EC, there is a need for including them in monitoring and assessment schemes as they may considerably contribute to the pollution load of the river basin. Finally, guidelines to manage these wastes through technologies that minimise their environmental impact and lead to a sustainable use of resources are also critical.

Keywords Benthic fauna, Ecological status, Effluents, Olive mills, Toxicity

---

I. Karaouzas (✉)
Institute of Marine Biological Resources and Inland Waters, Hellenic Centre for Marine Research, 46.7 km Athens-Sounio Av., 19013 Anavissos, Greece
e-mail: ikarz@hcmr.gr

N. Skoulikidis et al. (eds.), *The Rivers of Greece: Evolution, Current Status and Perspectives*, Hdb Env Chem (2018) 59: 169–204, DOI 10.1007/698_2016_453,

Contents

1 Introduction

Throughout the course of human history, the quality and quantity of water were crucial determinants of human health and the health of the Earth's ecosystems. The dramatic yet continuous industrial and agricultural development of the past century has significantly degraded the environment and particularly the soil, lakes, rivers and other aquatic ecosystems. It is estimated that river ecosystems have deteriorated more than any other aquatic ecosystem [1, 2] mainly due to changes in land use, organic and chemical pollution (agrochemicals, solid and liquid industrial and municipal wastewaters), overexploitation of water resources (e.g. water abstraction, overfishing, sand and gravel extraction, etc.), reduction and deforestation of riparian vegetation and unintentional and intentional introduction of exotic and alien species. Pollution episodes are daily, and in many cases, their impact on ecosystems is unpredictable and terrifying.

Rivers play a key role in ecosystems and provide a series of ecosystem functions such as habitat and food source for a wide range of biological species and ecological refuge development. Historically, rivers accommodated communities by providing food and water and a medium for transport, recreation and tourism. Inevitably, many peri-urban and floodplain rivers draining from urban and agricultural areas have been affected significantly during the last decades and remain a sensitive issue in the agenda of river management authorities.

Agricultural industries (referred to as agroindustries hereafter) are major contributors to the worldwide industrial pollution problem. With the tremendous pace of technological development to cover the needs of population overgrowth, the amount and complexity of wastes generated by these industries and their management has been problematic. Now, agroindustries, more than any other industrial sector, require an appropriate approach for successful waste management. There is no wonder that until 2004, more than 1,000 references on the various treatment methods of olive mill wastewaters have been published worldwide [3], and that number has been constantly increasing. Agroindustries such as olive oil mills, fruit processing factories, cheese factories and dairy farms constitute one of the most important pillars of local economy for the Mediterranean countries, including Greece. Agroindustries processing agricultural raw materials such as fruit, vegetables and animal products produce millions of tons of wastewater and large amounts of by-products, which are left untreated or unexploited and end up in the environment. These industrial facilities are usually scattered throughout the countryside, and the raw materials processed are produced at a seasonal rate, thus resulting to wastes varying significantly during the year both in quantity and characteristics.

In this chapter, the environmental impacts of agroindustrial wastewater discharge on river ecosystems of Greece are reviewed and discussed, focusing especially on major industries such as (a) olive mills, (b) orange juice processing factories and (c) cheese processing factories. In addition, impacts from other agroindustries are briefly highlighted.

2 Olive Mill Wastewaters

2.1 *Current Production Trends*

Worldwide, olive cultivation has increased significantly due to population increase and cultivation intensification using fertilisers, pesticides, irrigation of olive groves and new processing technologies of olives. In Greece, the number of olive trees was estimated to be around 75 million in 1961, while in 2003, their number reached 137 million, an 82.6% increase (Hellenic Ministry of Rural Development and Food). Olive oil production in 1961 was approximately 215 thousand tons, and in 2012, production reached 352 thousand tons, an increase of 64% (Fig. 1). Currently, there are about 14×10^7 olive trees and 450 approved olive mill establishments (Fig. 2), although the real number is estimated to be around 2,800 olive oil mills. Thirty percent (30%) of the olive mills are found in the Peloponnese, 24% in Crete, 9% in Attica, 7% in Western Greece, 7% in Central Greece, 9% in Macedonia and Thrace, 4% in the North Aegean, 4% in the Ionian, 3% in Thessaly and finally, 2% in Epirus and South Aegean, respectively.

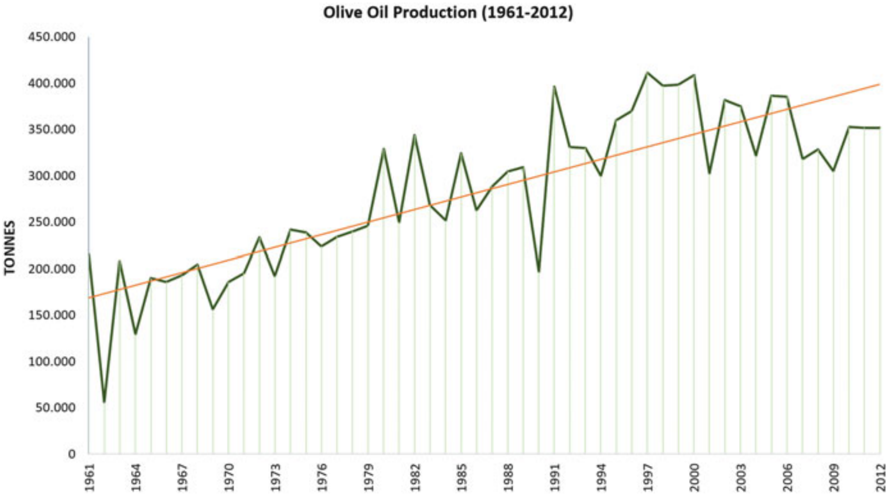


Fig. 1 Olive oil production (tons) in Greece from 1961 to 2012 according to FAO [4]

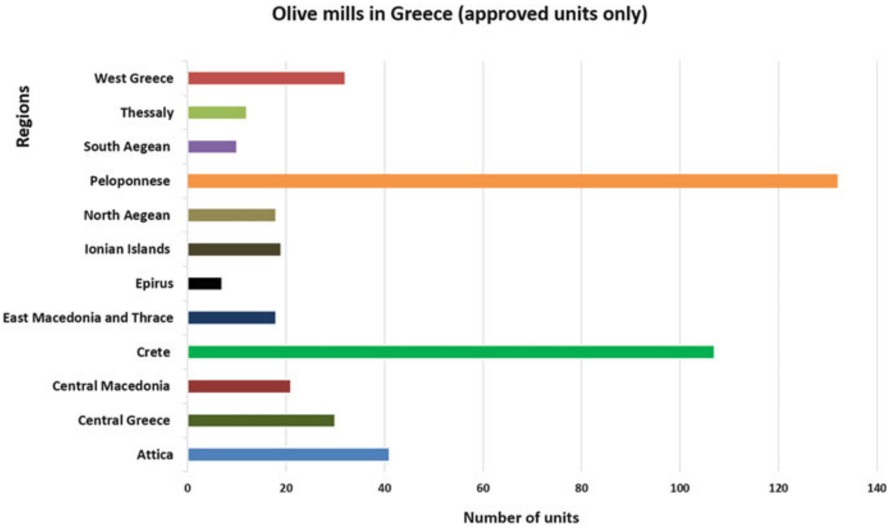


Fig. 2 Approved olive oil mill establishments for 2015 registered at the Greek Ministry of Rural Development and Food

2.2 *Polluting Capacity and Characteristics*

Olive mill wastewater (OMW) is one of the major and most challenging organic pollutants in olive oil production countries [5, 6]. OMW is the turbid liquid waste generated during the extraction of olive oil, where huge quantities of organic wastes are produced within a short period and usually lasts 3–5 months (November–

March). It is estimated that the volume of OMW produced annually in the Mediterranean region varies between 7×10^6 and 30×10^6 m^3 [7]. Despite the global spread of the olive tree, 95% of the production of olive oil (which yields about 2.5 million tons of olive oil per year) comes from the Mediterranean countries with Spain, Italy and Greece being the largest producers.

The milling process of olives generates about 50% of wastewater, 30% of solid residues and 20% of olive oil. OMW is easily fermentable and its characteristics are variable depending on the method of extraction, type of olive variety, soil and climatic conditions and cultivation methods. Typical OMW composition by weight is 83–94% water, 4–16% organic compounds and 0.4–2.5% mineral salts [8]. The wastewater arising from the milling process amounts to 0.5–1.5 m^3 per 1 ton of olives, depending on the process method [9, 10]. The high pollution ability of OMW is attributed to its remarkably high organic load (BOD: 25–100 g/l; COD: 45–220 g/l) and high content of phenolic compounds [10, 11], its acidity (pH 4–5) as well as the significant concentrations of magnesium, potassium and phosphate salts [12]. In addition, OMW contains many organic compounds such as lipids, sugars, organic acids, tannins, pectins and lignins that contribute to the increase of its organic load [8, 10]. Table 1 presents the major physical and chemical properties of OMW.

Although disposal of untreated OMW in aquatic systems is not allowed in Greece, it is estimated that approximately 1.5 million tons of OMW are disposed of every year in rivers, streams (Fig. 3 and 4), lakes and even in the sea [5]. The effective treatment of OMW requires expensive and advanced technologies that most olive mills lack. The usual treatment and disposal practice followed in Greece involves neutralisation with lime and disposal in evaporation ponds/lagoons. Disposal of OMW causes significant environmental pollution with unforeseeable effects on the quality of soil, surface and groundwater [17, 18] and poses a serious risk to aquatic and terrestrial biota and subsequently to the health of corresponding ecosystems.

2.3 *Toxicity of Olive Mill Wastewaters*

OMW and its polyphenolic fraction can be toxic to aquatic organisms [14, 15, 17, 19, 20], to bacteria and yeast [21] and to seed germination [22]. Moreover, it has been shown to affect the physical and chemical properties of the soil and its microbial community [7, 18, 23, 24], while several studies have verified its phytotoxic effects and antimicrobial activity [25]. Finally, OMW can be toxic to anaerobic bacteria which may inhibit conventional secondary and anaerobic treatments in municipal treatment plants [26].

Many related toxicological studies have evaluated the toxicity of the whole OMW effluent with standard toxicity test organisms such as *Daphnia pulex*, *Daphnia magna* and *Thamnocephalus platyurus* [15, 17, 20, 25] or with the luminescent bacteria *Vibrio fischeri* [27]. Paixão et al. [15] have shown that the LC_{50} acute toxicity of OMW can range from 1.08% to 6.83% for *Daphnia magna*,

Table 1 Physicochemical characteristics of OMW

Parameters	Units	Press mill	Three-phase centrifugation system
Density	(g/cm3)	1–1.2	1–1.1
Salinity	mmhos/cm	8–16	8–16
pH		4.2–5.3	4.6–5.2
Conductivity	mmhos/cm	12–18	8–16
Total solids	g/l	70–173	45–103
Total suspended solids	g/l	2–7	2.5–5
BOD_5	g/l	60–100	25–50
COD	g/l	65–190	45–110
Total phenols	g/l	12–19	6–10
Hydroxytyrosol	g/l	0.07–0.9	0.04–0.4
Phenolic acids	g/l	0.5–0.6	0.2–0.3
Tannins-Lignins	g/l	3–12	3–10
Pectins	g/l	2–5	1.5–3
Fats and oils	g/l	1.5–3	0.5–1.64
Total sugars	g/l	17–32	11–21
Glycerol	g/l	0.1	0.062
Organic acids	g/l	2–7	2–4
Polyalcohols	g/l	3–6	2–4
Total N (Kjeldahl)	g/l	1–1.5	0.7–0.9
Organic N	g/l	0.1–1.1	0.1–1
Total proteins	g/l	20–37	11–23
Ash	g/l	7–11	4–8
TOC	g/l	50–70	35–45
Total phosphorus as P_2O_5	g/l	0.5–0.9	0.5–0.6
Nitrate	mg/l	20–23	10–12
Chloride	mg/l	219.48	124
Sulphate	mg/l	75–115	52–75
Iron as FeO	mg/l	35–48	16–32
Potassium as K_2O	g/l	2–3	2–2.5
Sodium as Na_2O	mg/l	300–500	200–300
Calcium CaO	mg/l	350–380	120–270
Magnesium MgO	mg/l	74–200	48–50
Silicate (SiO_2)	mg/l	24–31	16–22
Manganese	mg/l	16–20	11–12
Zinc	mg/l	16–20	11–14
Copper	mg/l	8–10	6–9
Lead	mg/l	0.5–2	0.4–0.7
Cobalt	mg/l	0.2–0.9	0.1–0.5
Nickel	mg/l	0.5–1.5	0.3–1.5

Data assembled from: [8, 10, 13–16]

Fig. 3 Illegal OMW discharge in Skatias stream of the Evrotas river basin (Peloponnese, S. Greece)

Fig. 4 OMW discharge through a pipeline in Skatias stream (Evrotas River, (Peloponnese, S. Greece)

0.73% to 12.54% for *Thamnocephalus platyurus* and 0.16% to 1.24% for the luminescent bacteria *Vibrio fischeri*. Similarly, Rouvalis et al. [20] also showed that the acute toxicity of OMW could vary from 1.7% to 12.4% for *Daphnia pulex* and 3.3% to 8.9% for *Thamnocephalus platyurus*.

More recently, the toxicity of the whole OMW effluent to aquatic macroinvertebrates has also been studied [14, 28]. The 24-h LC_{50} values of OMW range from 2.64% to 3.36% for *Gammarus pulex* and 3.62% to 3.88% for *Hydropsyche peristerica* [19]. Based on a five-class hazard classification system developed by Persoone et al. [29], for wastewaters discharged into the aquatic environment, olive mill wastewaters are classified as highly toxic [19]. OMW concentrations can also be lethal to crustaceans even at lower volumes. The 24-h LC_{50} value for the Palaemonidae species of Pamisos River in South Peloponnesus was 0.7% [33]. Table 2 summarises all known toxicological studies of OMW that have been conducted in Greece.

Table 2 Toxicological studies conducted in Greece evaluating OMW toxicity to plant and animal species

OMW Origin	Taxonomic group	Test organism	Endpoint	Toxicity values	References
Chania	Bacteria	*Vibrio fischeri*	Mortality (EC_{50})	OMW	[27]
				15 min – 0.47% (Conf. range: 0.37–0.6%)	
				30 min – 0.5% (Conf. range: 0.4–0.64%)	
				Electrochemically oxidised (60 min) OMW	
				15 min – 1.36% (Conf. range: 1.17–1.58%)	
				30 min – 1.13% (Conf. range: 1–1.29%)	
				Electrochemically oxidised (120 min) OMW	
				15 min – 0.06% (Conf. range: 0.05–0.07%)	
				30 min – 0.06% (Conf. range: 0.05–0.07%)	
				Electrochemically oxidised (180 min) OMW	
				15 min – 0.22% (Conf. range: 0.13–0.38%)	
				30 min – 0.04% (Conf. range: 0.01–0.14%)	
				(toxicity increased due to the formation of organochlorinated by-products)	
Crete	Bacteria	*Vibrio fischeri*	Mortality (EC_{50})	5 min – Untreated OMW 0.219% (±0.044)	[30]
				Treated OMW (1 cycle of treatment): 1.612 ± 0.687%	
				Treated OMW (2 cycles of treatment): 4.606 ± 1.51%	
				15 min – Untreated OMW: 0.187 ± 0.035%	
				Treated OMW (1 cycle of treatment): 1.361 ± 0.528%	
				Treated OMW (2 cycles of treatment): 4.374 ± 1.54%	

Crete	Crustaceans	*Artemia franciscana*	Mortality (EC_{50}) 24–48 h	Slight toxicity of OMW was observed at high dilutions (1.25 and 2.5% v/v). 100% mortality occurred following a 24 h exposure to OMW at 20% v/v and a 48 h exposure OMW at 10% v/v	[30]
Magnesia	Crustaceans	*Artemia franciscana* (Artoxkit M test)	Mortality (EC_{50}) 24 h	12 samples of seawater: < 10%, with the exception of two sites with toxicity values of 47 and 23% (estuarine areas with low salinity waters and possibly unfavourable conditions for the survival of *A. franciscana*)	[31]
Fthiotida and East Attica	Crustaceans	*Artemia* sp.	Mortality (LC_{50}) 24 h	Untreated OMW: 4.5%	[25]
				Biotreated OMW with *Pleurotus ostreatus*: 12.5%	
Fthiotida and East Attica	Crustaceans	*Daphnia magna*	Mortality (LC_{50})	48 h – Untreated OMW: 2.5% (was not affected by the treatment with *Pleurotus* strains, mortality of the population reached zero at very low OMWs concentration <2%)	[25]
Chania	Crustaceans	*Daphnia magna* (Daphtoxkit F ™ magna)	Immobilisation 24 h	OMW: 5% (Conf. range: 2.5–7.5%)	[27]
				Electrochemically oxidised OMW (60 min): 50%	
				Electrochemically oxidised OMW (120 min): 40%	
				Electrochemically oxidised OMW (180 min): 25%	
				(Increased toxicity after oxidation of OMW)	
Magnesia	Crustaceans	*Daphnia magna* (Daphtoxkit F™ magna)	Mortality (EC_{50}) 24 h	24 h – 12 samples of freshwater: < 5%, with the exception of one site (high amount of agrochemicals draining from the neighbouring farmlands) with toxicity value of 15%	[31]
Achaia (OMW) and Kalamata (OMW)	Crustaceans	*Daphnia pulex* (Daphtoxkit F™ pulex)	Mortality (LC_{50}) 24–48 h	24 h – anaerobically treated OMW: 7.64% ± 4.5, anaerobically treated OMW: 8.83% ± 6.5 (18% of the samples toxic and 82% very toxic)	[32]
				48 h – anaerobically treated OMW: 4.38% ± 2.6, 4.38% ± 2.6, anaerobically treated OMW: 4.48% ± 2.9 (5% of the samples toxic and 95% very toxic)	

(continued)

Table 2 (continued)

OMW Origin	Taxonomic group	Test organism	Endpoint	Toxicity values	References
Achaia	Crustaceans	*Daphnia pulex* (Daphtoxkit F^{TM} pulex)	Mortality (EC_{50}) 48 h	OMW: 1.7–12.4%	[20]
				Toxic Unites (TU) = 8.1–59.2, toxic to very toxic	
Laconia (Evrotas River)	Crustaceans	*Gammarus pulex*	Mortality (LC_{50}) 24 h	OMW: 2.64–3.36%	[19]
				TU = 29.76–37.88	
				Class IV – high acute toxicity	
Fthiotida and East Attica	Crustaceans	*Heterocypris incongruens*	Mortality,% Growth Inhibition	No effect on growth inhibition between untreated and treated OMW. Toxicity slightly decreased after treatment. The concentration of OMW that caused 50% mortality increased from 3.7% (in the untreated OMW) to 5% (in the treated OMW)	[25]
Messinian rivers	Crustaceans	Palaemonidae shrimp	Mortality (LC_{50}) 24 h	OMW: 0.7%	[33, 34]
				TU of OMW: 143 – Class V, Very High Acute Toxicity	
Achaia	Crustaceans	*Thamnocephalus platyurus* larvae (Thamnotoxkit F)	Mortality (LC_{50}) 24 h	Untreated OMW: $0.94 \pm 0.66\%$, TU = 106.4	[26]
				Anaerobically treated OMW: $1.6 \pm 0.48\%$, , TU = 62.5	
				(TU > 100 : extremely toxic, TU = 11–100 : very toxic)	
Achaia (OMW) and Kalamata (OMW)	Crustaceans	*Thamnocephalus platyurus* larvae (Thamnotoxkit F)	Mortality (LC_{50}) 24 h	Anaerobically treated OMW: 1.77% ± 0.8	[32]
				Anaerobically treated OMW: $2.32\% \pm 1$	
				(8% of the samples are extremely toxic and 82% very toxic)	
Achaia	Crustaceans	*Thamnocephalus platyurus* larvae (Thamnotoxkit F)	Mortality (LC_{50}) 24 h	OMW: 3.3–8.9%	[20]
				TU = 11.1–30.4, very toxic	

Laconia (Evrotas River)	Trichoptera	*Hydropsyche peristerica*	Mortality (LC_{50}) 24 h	OMW: 3.62 (Conf. Limits: 3.19–4.11%) – 3.88% (Conf. Limits: 3.45–4.37%)	[14]
				TU = 25.77 (Conf. Limits: 22.88–28.99) – 27.62 (Conf. Limits: 24.33–31.35)	
				Class IV – high acute toxicity	
Patras	Mollusks	*Mytilus galloprovincialis*	Stress indices in tissues	Mussels exposed to either 0.1 or 0.01% (v/v) OMW for 5 days. Decreased neutral red retention (NRR) assay time values, inhibition of acetylcholinesterase (AChE) activity. Increase of micronucleus (MN) frequency and DNA damage were detected in haemolymph/haemocytes and gills	[35]
Messenia	Fungi	*Pleurotus* strains	Biomass	OMW toxicity as evaluated by the mycelium growth of *Pleurotus* strains was influenced significantly by the phenolic content of OMW samples obtained during three successive crop years; in contrast, the olives harvest period did not affect *Pleurotus* bio-mass production	[36]
Achaia	Fish larvae	Zebrafish *Danio rerio* embryo	Mortality (LC_{50}) 24–48 h	24 h – Untreated OMW: 0.43% (Std = 0.19), TU = 231.4, extremely toxic	[26]
				Anaerobically treated OMW: 2.33% Std=0.77), TU = 42.9, very toxic	
				48 h – Untreated OMW**:** 0.33% (Std=0.15), TU = 304.4, extremely toxic	
				Anaerobically treated OMW**:** 1.85% (Std=0.57), TU = 54, very toxic	
				(TU > 100 : extremely toxic, TU = 11–100 : very toxic)	

(continued)

Table 2 (continued)

OMW Origin	Taxonomic group	Test organism	Endpoint	Toxicity values	References
Achaia (OMW) and Kalamata (OMW)	Fish larvae	Zebrafish *Danio rerio* embryo	Mortality (LC_{50}) 24–48 h	24 h – anaerobically treated OMW: 1.99% ±1.1, anaerobically treated OMSW: 2.09% ±0.7	[32]
				48 h – anaerobically treated OMW: 1.52% ±0.8, anaerobically treated OMSW: 1.63% ± 0.8	
				(26% of the samples are extremely toxic and 74% very toxic)	
Crete	Mammals	Wistar rats	Mortality, body weight, signs of toxicity such as tremor, convulsion, salivation, etc.	Oral administration of 2,000 mg OMW per kg body weight. No mortality, no signs of toxicity or abnormal behaviour during the 14-day observation period, normal body weight development	[30]
Fthiotida and East Attica	Plants	*Lepidium sativum*	Seed germination	Untreated OMW – Germination Index (G.I.): 0, Biotreated OMW with *Pleurotus ostreatus* strains LGAM P113 and P115 – increased G.I., toxicity significantly decreased	[25]
Chania	Plants	Lettuce	Seed germination	OMW is strongly phytotoxic (even at 1:4 dilution) and completely hinders plant growth	[37]
Kalamata	Plants	*Lycopersicon esculentum* Mill.	Plant growth	Highly significant reduction of growth observed. Root was more sensitive to OMW than the upper parts of the tomato plant which may be because the root face OMW toxicity directly, while toxicity to other parts is indirect	[38]
Kalamata	Plants	*Spinacia oleracea* L. cv Virofly	Seed germination, plant growth, shoot and root elongation and biomass	1:10 diluted OMW suppressed seed germination by 30%	[39]
				1:20 diluted OMW suppressed seed germination by 22% ($p < 0.01$)	

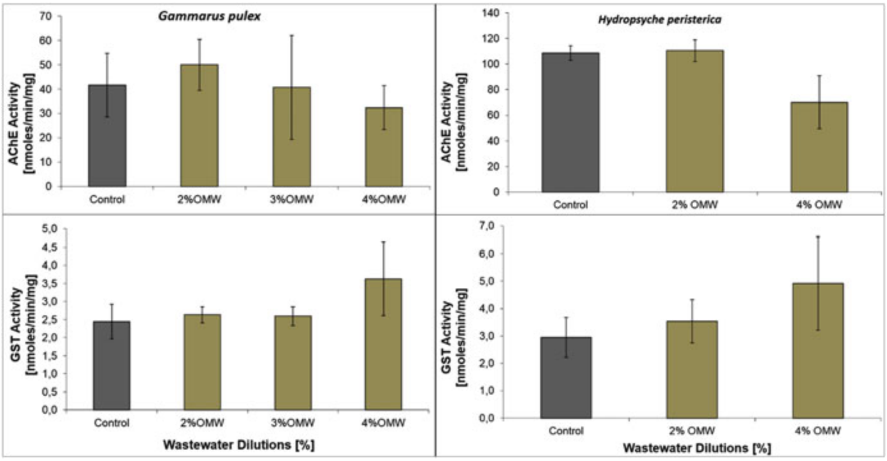


Fig. 5 AChE and GST activities of *G. pulex* (*left*) and *H. peristerica* (*right*) exposed to olive mill wastewaters for 24 h. Results are expressed as the mean ± SD per wastewater concentration samples

Sublethal concentrations of OMW can also cause damage at lower levels of biological organisation [14, 28]. For example, the enzyme activities of the caddisfly *Hydropsyche peristerica* and amphipod *Gammarus pulex* were affected when exposed to OMW [19]. The acetylcholinesterase (AChE) activity of *H. peristerica* and *G. pulex* decreased after 24 h of exposure (Fig. 5). In contrast, the glutathione S-transferase activity of the two species has been shown to increase as OMW concentration increases (Fig. 5). Inhibition of AChE was also observed in the mussel *Mytilus galloprovincialis* when exposed to either 0.1 or 0.01% (v/v) OMW for 5 days [35]. Specifically, decreased neutral red retention (NRR) assay time values, inhibition of acetylcholinesterase (AChE) activity, as well as a significant increase of micronucleus (MN) frequency and DNA damage were detected in haemolymph/haemocytes and gills, compared with values measured in tissues of control mussels [35].

2.4 Effects on Water Quality, Aquatic Organisms and Ecological Status

Olive mill wastewaters are being discharged, untreated or partially treated in hundreds of torrents and streams throughout the country. The most visible effect of OMW pollution is the discolouring of surface waters, which is attributed to the oxidation and subsequent polymerisation of tannins that give dark-coloured polyphenols [13]. The main cause of the problem is the very high organic content, which is not easily biodegradable, while high concentrations of polyphenolic compounds result in toxicity and environmental degradation. Most mills are family businesses of small capacity which cannot afford the cost of installing treatment systems, ending up in disposing the wastewater in adjacent water bodies. In fact, the main recipients of wastewaters in Greece are streams and torrents (58.3%), soil (19.8%), rivers (6%), water (5.3%) and lakes (0.038%) [40]. There have been numerous studies on the effects of pollution in Greece's running waters, resulting from the synergistic effect of multiple stressors, such as pesticides, fertilisers and hydromorphological degradation; however, the pure effects of OMW on the aquatic biota and ecological status of stream ecosystems have been poorly investigated (Table 3).

The first one dates back to 1993, where Voreadou [44] within the context of her doctorate thesis studied the impacts of OMW in several streams of Crete. The results of her study showed a dramatic decline of the benthic macroinvertebrate community during wastewater discharge, while the intensity of the effects was proportional to the volume and duration of water in the stream bed. In streams receiving OMW with high water velocity that retained water for 7–8 months, species richness declined by 41%, while in streams with less water supply and flow duration, species richness loss reached approximately 71% [45]. Voreadou [44], apart from the phenol toxicity capacity of OMW, also attributed the reduction of biodiversity to the formation of a greasy layer on the water surface from the lipid content of the wastewater, thus preventing the entry of light and oxygen and the accumulation of solid components in the stream bottom that may enter the body of aquatic organisms.

Recently, the effects of OMW on the stream macroinvertebrates, water quality and river ecological status were thoroughly and systematically studied in the Evrotas river basin in South Peloponnese [14, 19, 28, 42]. Benthic macroinvertebrates and environmental parameters were monitored for two years, thus following the biennial cycle of olive growth and production and hydrological variation (dry – wet years) in order to assess spatial and temporal responses of stream fauna to high and low OMW yield years. Furthermore, two different hydrologic years (wet and dry year) were covered during the two-year monitoring period, thus allowing evaluation of hydrologic regime variation to OMW pollution intensity and effects.

The results of these studies revealed the spatial and temporal structural deterioration of the aquatic community due to OMW discharge with consequent reduction of the river capacity for reducing the effects of polluting substances through internal mechanisms of self-purification. OMW, even highly diluted, had significant impacts on the aquatic fauna and the ecological status of the Evrotas River. The vast majority of macroinvertebrate taxa were eliminated, and only a few tolerant Diptera species (i.e. Chironomidae, Simuliidae, Syrphidae) survived with very limited abundances (1–4 individuals/1.25 m^2). Macroinvertebrate assemblages downstream the OMW outlets were dominated by Diptera species, whereas Ephemeroptera, Plecoptera and Trichoptera (EPT) were almost depleted during and after the OMW discharge period.

Overall, the effects of OMW on water chemistry were more pronounced on the second year of the sampling campaign due to the higher olive fruit production that

Table 3 Monitoring and assessment studies conducted in Greece evaluating OMW effects to running waters

River basin	Stream name	Investigated topic	Endpoint (State/Effect)	References
Pamisos, Nedon, Aris, Belikas and Epis	Pamisos, Nedon, Aris, Belikas and Epis and their estuaries	Effects of OMW on water quality	Water quality (Downgraded especially in November and December. Elevated levels of phenols, high concentrations of ammonium and inorganic phosphorus)	[34]
Epis	Epis and its estuary	Effects of OMW on water quality	Physicochemical quality (Increased values of Mn, Cu, Ni, phenols, ammonium, nitrates)	[41]
Evrotas	Kotitsanis, Vordoniatis, Yerakaris and Skatias tributaries	Effects of OMW on benthic macroinvertebrates and ecological status	Water pollution (increased COD, BOD_5, TSS, chloride, phenols, sewage bacteria levels, decreased O_2 concentrations)	[42]
			Macroinvertebrate assemblages (decreased number and abundance of taxa, degraded biocommunity structure)	
			Biological quality and ecological status (downgraded from good and high to moderate and bad)	
Evrotas	Kotitsanis, Vordoniatis, Yerakaris and Skatias tributaries	Effects of OMW on small streams	Physicochemical quality (good to moderate)	[42]
			Biological quality and ecological status (moderate to bad)	
			Water pollution (high levels of BOD_5, COD, TSS and phenols, low DO)	
Evrotas	Evrotas River	Effects of OMW on water and effects of disposing OMW	Water physicochemical quality (low levels of COD, phenols and nutrient levels in Evrotas river. Increased phenols in Skoura and Vrontamas station due to the olive mills)	[43]
			Attenuation capacity of river sediments (High	

(continued)

Table 3 (continued)

River basin	Stream name	Investigated topic	Endpoint (State/Effect)	References
			attenuation capacity. Phenols were reduced from 2.0 to 1.0 mg/l and COD from 30.3 to 6.3 mg/l in 68 days)	
			Soil physicochemical quality after irrigation with treated OMW (increased conductivity, pH, nitrogen, nitrate-N and organic matter, lower ammonia-N)	
			Groundwater quality (increased phenolic compounds, ammonia, TOC and COD. Leaching of loads from the surface to the groundwater)	
Aposelemis	Prinopotamos	Effects of OMW on water quality and benthic fauna	Benthic macroinvertebrate community loss during wastewater discharge. Intensity of the effects was proportional to the volume and duration of water in the stream bed. Streams receiving OMW with high water velocity that retained water for 7–8 months, species richness declined to 41%, while in streams with less water supply and flow duration, species richness loss reached approximately 71%	[44]

yielded a greater quantity of wastewater. During the OMW discharge period, BOD_5, COD and TSS were extremely high, causing a significant decrease in dissolved oxygen concentrations and creating anoxic conditions in many cases. A significant increase in chloride and total phenols concentration was also observed in the downstream sites during the wastewater discharge period as well as a marked increase in nutrients [42]. Mean concentrations of COD, BOD_5, total phenols, total suspended solids (TSS) and chloride were higher in the sites receiving OMW, while sewage bacteria flourished as a result of OMW residue on the stream substratum during the wastewater discharge period [42]. Dissolved oxygen concentration showed no marked variation among periods in the upstream sites in contrast to

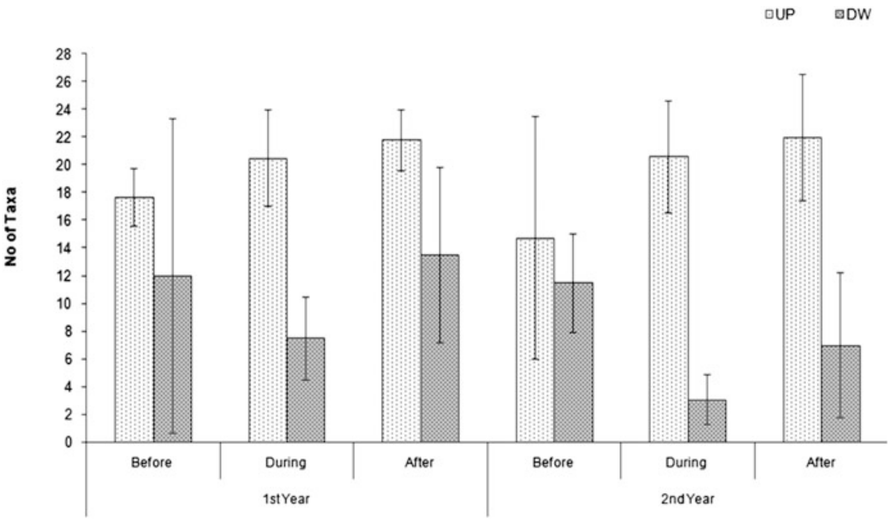


Fig. 6 Mean (± SD) number of taxa before, during and after the OMW discharge period for the two-year sampling campaign. *UP* upstream sites; *DW* downstream sites ([42], with permission)

the downstream sites, where oxygen concentration decreased during and after the wastewater discharge period, especially in the second year of sampling. Total phenols were detected only during the wastewater discharge period and were significantly higher in the second sampling year compared to the previous one.

Species richness downstream the OMW outlets was markedly lower than upstream of the olive mills (Figs. 6, 7, and 8). During the wastewater discharge period, the number and abundance of taxa were significantly decreased; the effects during year two being more pronounced due to the prolonged drought in the years 2006 and 2007 (Fig. 6). Upstream sites that were used as control presented good and high ecological status, whereas the ecological status of the sites affected from OMW pollution ranged from moderate to bad. Effects were more pronounced at lowland intermittent streams (Fig. 8), thus showing that intermittency and prolonged drought in combination with wastewater discharge significantly affect stream fauna and ecological status. Stream typology (i.e. slope, altitude) and hydrology of the stream site (i.e. perennial or intermittent) and the intensity and volume of the wastewater were the most important determinants of self-purification processes [42].

A study conducted in the Epis River in Messenia (Peloponnese, S. Greece) by Anastasopoulou et al. [41] revealed high concentrations of phenols (36.1–178 mg/l), ammonium (7.3–9.5 mmol/l), phosphate (6.1–7.5 mmo/l), COD (53.4 g/l) and certain heavy metals especially during December when the olive oil production reaches a peak in the area. Increased values of Mn, Cu and Ni were recorded in the river water, while the calculation of the sediment enrichment factor confirmed the ecosystem's deterioration due to these trace metals. Increased phenol concentrations were also detected in the Messenian Gulf during the olive-harvesting period

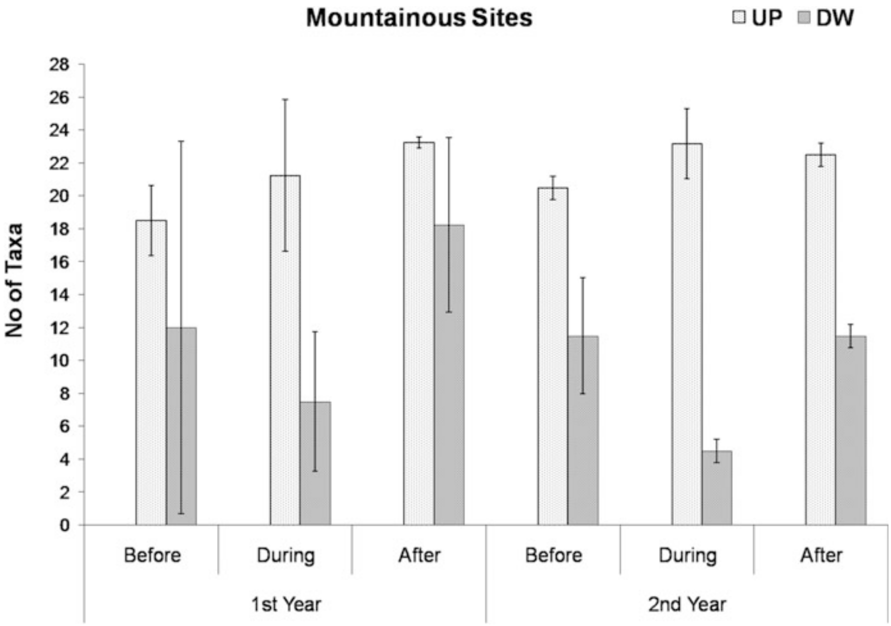


Fig. 7 Mean (± SD) number of taxa upstream and downstream the OMW outlet in mountainous (permanent) sites ([42], with permission)

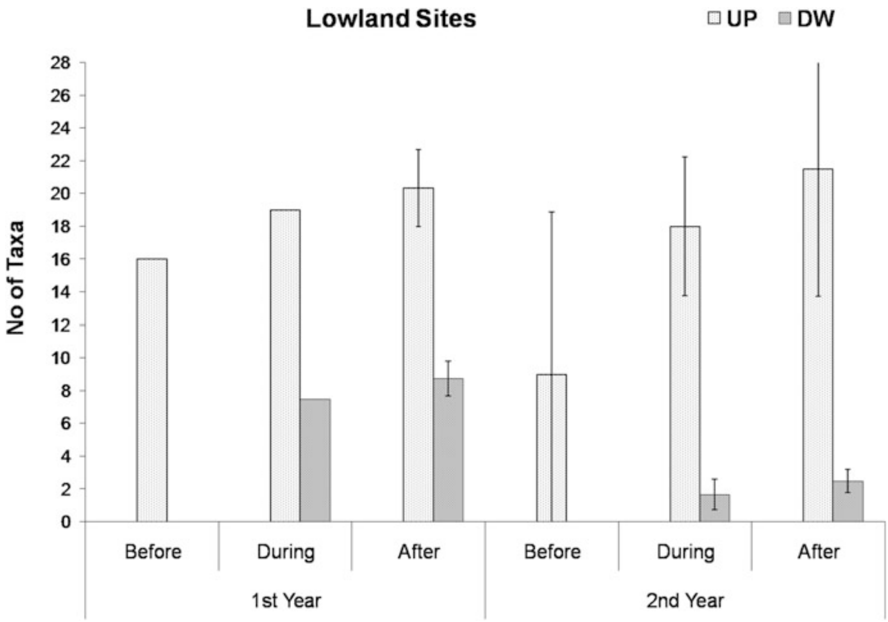


Fig. 8 Mean (±SD) number of taxa upstream and downstream the OMW outlet in lowland (intermittent) sites ([42], with permission)

(96–207 ppb). Concerning the concentration of heavy metals, high values of Fe (515 ppb) and Mn (486 ppb) were detected in the water body of Epis before its estuaries [41]. Before the production period, the concentration of these trace metals were found at much lower levels (48.9 and 118 ppb, respectively). Both zinc and lead's concentrations did not appear to differ greatly before and during the production period. According to the Greek Nutrient Classification System [46], the quality of all the sites assessed before the olive oil production period of 2008–2009 was classified as high. During the production period, the physicochemical quality of the sites downstream the olive mills varied from moderate to poor. Based on benthic macroinvertebrate fauna, the biological quality of the Epis River ranged from bad to moderate [47].

Another study conducted in the region of Messenia from 2008–2011 [34], in which several rivers and streams that discharge into the coastal zone of the Messenian gulf were included, showed that OMW have deteriorated the water quality of the gulf. The studied rivers were classified as good or moderate, and in some cases, poor, whereas the sites at the coastal zone of the Messenian Gulf were characterised as good or moderate [34]. Five months after the oil-productive period (May 2011), water quality has not been recovered due to OMW. This is also supported by the biological results (macroinvertebrates) in the studied rivers influenced by OMW, which were obtained in the framework of a monitoring programme that was carried out at the Prefecture of Messenia during 2011–2014 [48]. According to the STAR_ICMi [49] and BMWP [50] biological indices, the ecological status at most of the sampling sites of Pamisos River, Epis River, Belikas River and Aris River was classified as good or moderate during the summer period, with no influence of OMW, and as poor or bad during the wet (olive oil production) period [48].

Increased nutrient and metal concentrations are also reported from experiments carried out in evaporation lagoons in order to test for changes in the chemical properties of the soil [51]. Disposal of untreated OMW at evaporation lagoons without using protective materials (e.g. impermeable membranes) resulted in significant changes in soil chemical properties. Soil samples collected one month after the completion of waste disposal were characterised by enhanced content in nitrogen, organic matter, exchangeable K, Mg, cation exchange capacity, available Mn and Fe as well as increased electrical conductivity and decreased $CaCO_3$ [51]. Changes in soil properties depended on depth and distance from the disposal lagoon.

Although not carried out in running waters, a study performed by the Institute of Infectious and Parasitic Diseases of the Centre of Veterinary Institutions of Thessaloniki [98] on the effects of OMW in fish farms in the Gulf of Amvrakikos surfaced conclusions already known. Specifically, fish deaths occurred in aquaculture due to (a) non-water-soluble OMW components superimposed on the gills of the fish thus blocking respiration, (b) pH decrease of seawater at values well below 8 (which is the normal value in the region), (c) high levels of BOD and COD leading to anoxic conditions and (d) weakening of the organism, making them susceptible to microbial infections.

2.5 Current Legislation

The disposal of OMW, on both freshwater bodies and soils, may also affect groundwater quality [52], especially in calcareous rocks, which have high permeability. In Greece, according to the law Y2/2600/2001, the limit for the content of phenolic compounds in drinking water is 0.5 μg/l. The concentration of phenols in OMW usually ranges from 0.1 to 0.5 mg/l, which certifies that the direct disposal of wastewater into water bodies is hazardous. The acceptable limits designated for European countries for the disposal of OMW in various recipients are presented in Table 4.

Until today, there are no specific rules for the treatment, management and disposal of OMWs in surface water bodies. The Ministry of Development, Competitiveness, Infrastructure, Transport and Networks and the Ministry of the Environment, Energy and Climate Change (YPEKA) have published a guideline on the management of OMW that is included in Category B of the Ministerial Decision (MD) 1958/2012 (Government Gazette B21/13-01-2012). The guidelines set for the application of the term E3 of the Common Ministerial Decision (CMD) Φ15/4187/266/2012 (Government Gazette B'1275/11-04-2012) on standard environmental commitments for certain industrial activities refers to pretreatment methods so as to avoid the direct discharge of wastewaters to water recipients. With the 191645/03-12-2013 Circular, the Secretariat for Water of the Ministry of Environment, Energy and Climate Change states that within the implementation of the measures of the Basin Management Plans of the water districts in the country, they will proceed to the modernisation of waste management legislation with the issue of a Common Ministerial Decision (CMD). The new CMD will replace the articles of the Sanitary Provision EIB/221/1965 on the disposal of liquid waste into surface water bodies and will essentially abolish it. Until the adoption of this new CMD, the decisions of the regional units should be followed.

Table 4 Acceptable levels on the disposal of OMW in water bodies in different European countries

Parameters [mg/l]	Disposal in surface waters			Disposal at sea		Disposal at sewage systems		
	Greece	Italy	Croatia	Greece	Croatia	Greece	Italy	Croatia
pH	6–9	5.5–9.5	6.5–8	6–9	6.5–8	6–9	5.5–9.5	5–9.5
BOD	40	≤40	25	40	25	500	≤250	250
COD	120	≤160	125	120	125	1,000	≤500	700
Total suspended solids	40	≤80	35	50	35	500	≤200	80
Lipids and oils	5	–	25	5	25	40	100	–
Total phenols	0.5	≤0.5	0.1	0.5	0.1	5	≤1	10

Source: IMPEL Olive Oil Project 2003

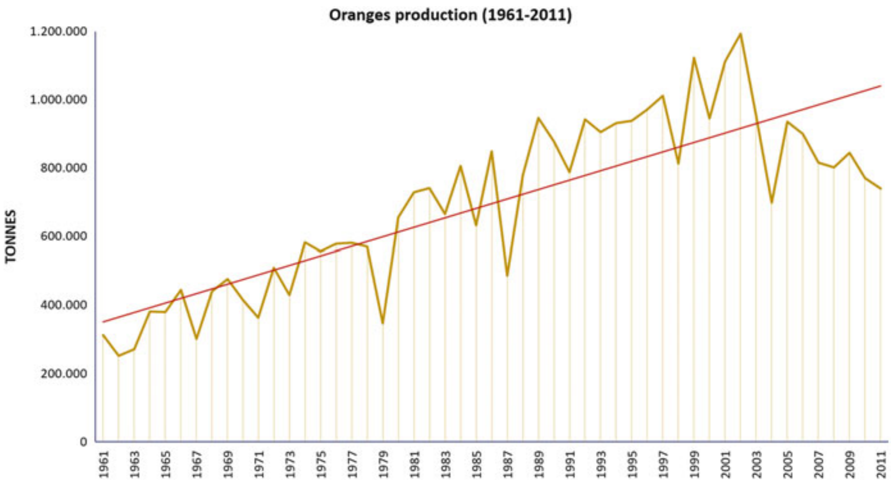


Fig. 9 Orange production (tons) in Greece from 1961 to 2011 according to the FAO (World Food and Agriculture Organization)

3 Orange Juice Processing Wastewaters (OJPW)

3.1 Current Production Trends

According to the Hellenic Ministry of Rural Development and Food, the total production of citrus fruit in Greece is about 1 million tons per year, of which only about a third is destined for juicing. The citrus fruit-cultivated area in Greece is estimated at approximately 53,000 hectares. Of these, 40,000 hectares are oranges [4]. The cultivated land and production of oranges have increased significantly in recent decades. In 1961, there were 17,700 hectares of orange trees that yielded 321,000 tons of oranges, and today, the cultivated area is about 40,000 hectares producing about 900,000 tons of oranges (Fig. 9). From this quantity, 34% of the fruit is produced in the prefecture of Argolida, 23% in Laconia, 17.5% in Arta, 9% in Chania and Crete, 5% in Ilia, and 3% in Etoloakarnania and Corinthia, respectively. The Greek citrus processing industries are mainly located in the regions of Argolida, Arta, Laconia and Crete.

3.2 Polluting Capacity and Characteristics

The processing of orange fruit gives about 45% fresh juice and 50% solids that consist of the pulp and peel of the fruit. The remaining 5% consists of a collection of cells, essential oils and limonene. A fraction of the juice and pulp are various compounds of flavonoids, such as hesperidin, neohesperidin, rutin, narirutin, naringin and nobiletin [53]. Orange juice production gives about 70% waste, of which 75–80% is solid and 20–25% liquid. Solid waste is composed of the peel and pulp, while the effluent comes from the washing of fruits and the production of various by-products. It is estimated that one ton of oranges produces 1.5 million litres of wastewater. In most citrus juice plants, the citrus wastewater undergoes biological treatment. The solid waste is usually transported to a landfill. However, disposal of waste in illegal dumps and steep cliffs, as well as the uncontrolled disposal of solid waste (pulp and shredded stems) in streams and rivers after being mixed with the wastewater is frequent (Figs. 10 and 11).

The composition and physicochemical characteristics of OJPW vary greatly depending on the variety, the maturity of the orange fruit and the production

Fig. 10 Illegal OJPW discharge in Tyflo stream of the Evrotas river basin, Peloponnese, S. Greece

Fig. 11 OJPW discharge in the Tyflo stream (Evrotas River, Peloponnese, S. Greece)

conditions. Wastewaters generated from orange juice production have high organic load (BOD: 20–1,400 mg/l; COD: 100–2,000 mg/l) and can be toxic due to the high concentration of organics, including terpene-containing oils and flavonoids [54]. The complex and insoluble carbohydrates, proteins, fibres, high nitrogen and sodium levels [55, 56] increase the organic load of OJPW and, in addition to the limonene levels (90–95% in citrus peel oil; 0.3%–0.8% in wastewater), decrease the effective treatment and disposal of the effluent [56]. Table 5 summarises the chemical and physicochemical composition of OJPW.

Table 5 Composition and physicochemical characteristics of OJPW

Parameters	Units	OJPW
pH		4–6.5
Acidity (citric acid)	g/l	0.1–0.2
Electrical conductivity	μS/cm	500–3,700
BOD_5	mg/l	409–4,000
COD	mg/l	435–13,650
Total solids	mg/l	640–840
Total suspended solids	mg/l	300–2,800
Total dissolved solids	mg/l	540
Alkalinity (CaCO3)	mg/l	800–815
Ash	mg/l	424
Total sugars	g/l	6–30
Total phenols	mg/l	1.5–8
Limonene	mg/l	50–200
D-Limonene	%	0.02–0.5%
Organics	%	94.7
Hesperidin	mg/l	1,000–3,000
Pectin	mg/l	1,200–9,000
Fats and oils	mg/l	2,045
Dry mass (DM)	g/kg	110
Proteins	g/kg	53.8
Fibres	g/kg	164
Nitrite (NO_2)	mg/l	1–3
Organic nitrogen [ON]	g/l	7.28
Potassium [K]	mg/l	1,578
Manganese [Mn]	mg/l	0.3–0.7
Iron [Fe]	mg/l	0.33–3.9
Total phosphorus [TP]	mg/l	188
Phosphorus [P]	mg/l	0.4–2.4
Calcium [Ca]	mg/l	30–60
Chloride [Cl]	mg/l	80–160
Sodium [Na]	mg/l	135–205

Sources: [55–60]

3.3 Toxicity, Effects on Water Quality, Aquatic Organisms and Ecological Status

Up to date, there is only one study available that documents the toxicity of OJPW on aquatic organisms [14, 19]. In that study, two test organisms were used for testing the acute toxicity of the wastewater: *Gammarus pulex* and *Hydropsyche peristerica*. Mortality for 50% of the amphipod *G. pulex* population occurred at 25.26% wastewater dilution concentration and 17.16% for *H. peristerica*. The latter showed to be more sensitive to OJPW toxicity than *G. pulex*. Based on the five-class hazard classification system used for wastewaters discharged into the aquatic environment [29], OJPW belongs to class III (acute toxicity).

The effects of OJPW were also evaluated at the molecular level of the two-test species by assessing changes in their AChE and GST enzyme activities [14, 28]. OJPW caused the decrease of AChE of *G. pulex* after 24 h of exposure (Fig. 12). Unlike the activity of AChE, the GST activity of *G. pulex* increased at higher concentrations of the effluent (Fig. 13). The same changes were also observed in the enzymatic activities of *H. peristerica* after 24 h of exposure. AChE concentration decreased at increasing concentrations of OJPW (Fig. 12), while activity of GST increased in conjunction with higher concentrations of the wastewater (Fig. 13).

As with olive mills, wastewaters of the citrus juice processing industry can have profound effects on freshwater ecosystems. Two streams of the Evrotas river basin

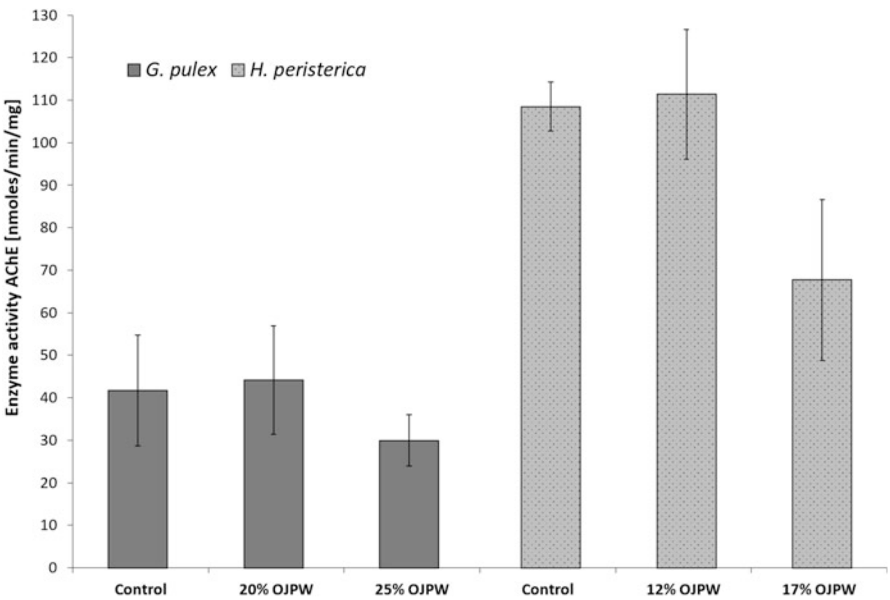


Fig. 12 AChE activities of *G. pulex* (*left*) and *H. peristerica* (*right*) exposed to orange juice processing wastewaters for 24 h. Results are expressed as the mean ± SD per wastewater concentration samples

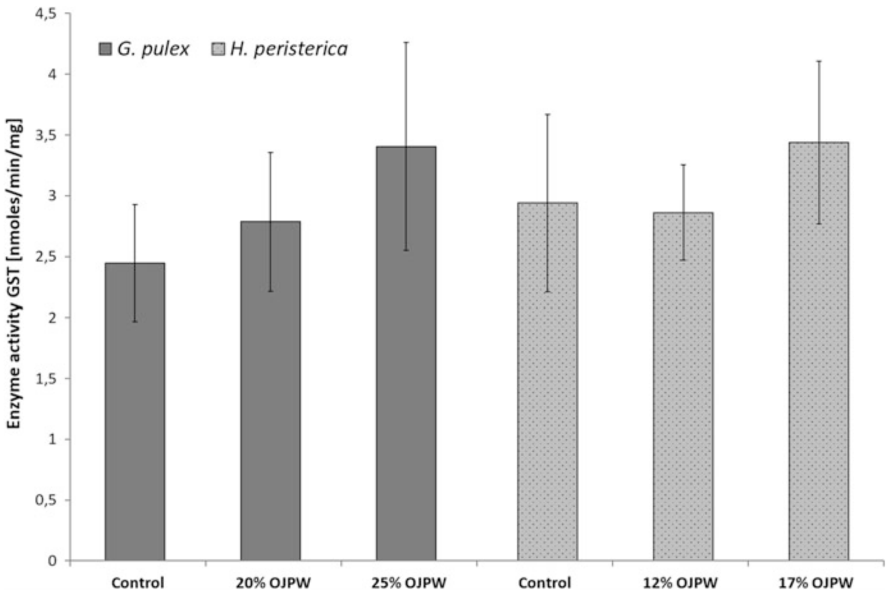


Fig. 13 GST activities of *G. pulex* (*left*) and *H. peristerica* (*right*) exposed to orange juice processing wastewaters for 24 h. Results are expressed as the mean ± SD per wastewater concentration samples

have been receiving untreated or partially treated wastewaters from two orange juice processing plants for many decades. Ecological quality monitoring and assessment carried out in these two streams revealed significant loss of the benthic fauna, since in almost all months of monitoring, only a few individuals of the Dipteran families of Chironomidae and Simuliidae were found [28]. The Tyflo stream flowing through the Riviotissa settlement on the suburbs of Sparta was represented solely by *Chironomus plumosus*-gr with very limited abundance (usually 1–3 individuals/1.25 m^2). Even months after the end of the wastewater discharge period, no recovery was observed in benthic fauna composition while the ecological status of the stream remained poor throughout the monitoring period. At the Mylopotamos stream in the Aghia Kyriaki settlement (3 km south of Sparta), the situation was the same as with the Tyflo stream, apart from a burst of *Chironomus plumosus*-gr. and Tubificidae worm abundances after the end of the wastewater discharge period [19, 28].

Apart from Evrotas, there are many more freshwater systems that receive wastewaters from fruit and vegetable juice processing units (oranges, peaches, apples, apricots, carrots, pomegranates, grapes, etc.) throughout Greece, such as Aliakmonas, Axios, Pinios, Louros, etc. Studies carried out in these systems involve their ecological status assessment and include a variety of stressors such as pesticides, nutrient pollution from fertilisers, organic pollution (olive mills, wastewater treatment plants, slaughterhouses), hydromorphological modifications, etc. [61–68].

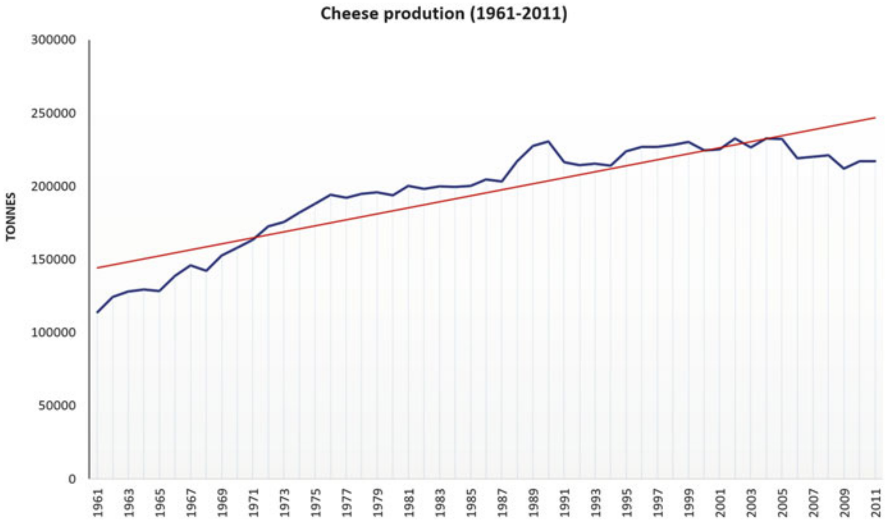


Fig. 14 Cheese production (tons) in Greece from 1961 to 2011 according to FAO (World Food and Agriculture Organization). Data include goat, sheep and cow cheese

4 Cheese Whey Wastewaters (CWW)

4.1 Current Production Trends

Cheese production in Greece is a traditional area of activity, as it has been reported by several historical sources as one of the main trade activities in ancient and more recent times. Over the years and with the assistance of financial institutions through granting investment incentives from the state (under development laws and EU regulations), the industry has made significant developments (Fig. 14). A key feature of the industry is the large number of industries, mainly primary production farms. The majority of these industries include, mainly, small size and capacity units at local level characterised by high dispersion and usually a lack of required modern mechanical equipment. Similar to olive mills, the exact number of cheese production units that currently operate in Greece is unknown. For example, in 2011, from the 96 registered production units of Crete, only 60 had operational permission, while the true number is speculated to be around 400 [99].

4.2 Polluting Capacity and Characteristics

The dairy industry is one of the main sources of industrial wastewater generation in Europe [69]. It is based on the processing and manufacturing of raw milk into products such as yogurt, butter, cheese and various types of desserts by means of several different processes, such as pasteurisation, coagulation, filtration, centrifugation, etc. Dairy factory wastewaters commonly contain milk, by-products of processing operations, cleaning products and various additives that may be used during the production [70]. The water requirement of a dairy plant for washing and cleaning operations corresponds to 2–5 l of water per litre of processed milk. The characteristics of dairy effluents may vary significantly, depending on the final products, system type and operation methods used in the manufacturing plant [71].

The cheese manufacturing industry generates three main types of effluents; cheese whey (resulting from cheese production), second cheese whey (resulting from cottage cheese production) and the washing water of pipelines, storage and tanks that generates a wastewater called cheese whey wastewater (CWW). The latter, also contains cheese whey and second cheese whey and is a strong organic and saline effluent whose characterisation and treatment have not been sufficiently addressed. CWW generation is roughly four times the volume of processed milk [71].

Cheese whey wastewater is white in colour and usually slightly alkaline in nature and becomes acidic quite rapidly due to the fermentation of milk sugar to lactic acid. It is characterised by an unpleasant odour of butyric acid, high organic content (COD up to 70 g/l, BOD up to 16 g/l) and relatively high levels of total suspended solids (up to 5 g/l) [72, 73]. Due to salt addition during the cheese production process, sodium and chloride levels are extremely high (2.1–2.8 g/l). The values reported in total nitrogen (0.5–10.8 mg/l) and phosphorus (6–280 mg/l) indicate a serious risk of receiving water eutrophication [71]. It also contains lactose, proteins and fats (45, 34 and 6 g/l, respectively) and has a high biodegradability index (BOD/COD $\approx$ 0.46–0.80) (Table 6) that suggests the suitability of biological process application [71].

Table 6 Composition and physicochemical characteristics of CWW

Parameters	Units	CWW
pH		4–8.7
Electrical conductivity [EC]	mS/cm	11–13
BOD_5	g/l	0.9–15
COD	g/l	0.8–77
Total solids [TS]	g/l	1–63.5
Total susp. solids [TSS]	g/l	0.25–5
Turbidity	NTU	1,300–2,000
Total organic carbon [TOC]	g/l	0.55–35
Total Kjeldahl nitrogen [TKN]	g/l	0.11–0.83
Total phosphorus [TP]	mg/l	6–280
Total nitrogen [TN]	mg/l	0.5–11
Fats and Oils	g/l	0.1–5.7
Proteins	g/l	1.88–9
Lactose	g/l	0.1–44
Chloride	g/l	2–2.5
N-NH4	mg/l	8–161

Sources: [71, 74, 75]

Fig. 15 CWW discharge in Voulgaris stream, Lesbos Island

If discharged untreated into the waterways (e.g. Fig. 15), CWW can cause serious environmental problems. Although cheese whey contains valuable fertiliser components such as nitrogen, phosphorus and potassium, application on land compromises the physical and chemical structure of the soil resulting to crop yield decline [76, 77] and reduces aquatic life by depleting the dissolved oxygen of the water [78, 79] and may eventually pollute the groundwater.

Several value-added products can be produced from cheese whey by using various fermentation processes in order to minimise the problems associated with its disposal and improve the economics of the dairy and food processing industry. Usually, the small and medium cheese factories are isolated from centralised wastewater treatment facilities and, in some cases, located next to ecologically sensitive areas, which may cause environmental risks. Land application is often the only practical option for wastewater disposal.

4.3 Toxicity, Effects on Water Quality, Aquatic Organisms and Ecological Status

Despite the fact that several methods for the treatment or utilisation of cheese whey wastewater have been proposed during the last 60 years, more than 50% of wastewater is discharged untreated to waterways [74]. According to statistical data of 2007, 7.5 million tons of CWW are produced every year in Greece [74]. The vast majority of these quantities are discharged untreated or partially treated into the environment, including soil and freshwaters.

Even though CWW discharge is among the main sources of organic pollution in Greek river ecosystems, its effects on aquatic ecosystems have overall been neglected. Many running waters throughout the country receive CWW, but up to date, effects only on the Vouraikos River in Peloponnese have been assessed

[80]. In that study, Karadima et al. [80] found that the ecological quality of the sites close to the cheese production factory ranged from moderate to bad and that there was a significant ecological risk for almost 15 km downstream of the point pollution source. Pollution-tolerant macroinvertebrate taxa such as Chironomidae, Tubificidae, Valvatidae and Lumbricullidae were abundant in the low-quality sites (close to the factory), which also presented low biodiversity values and low numbers of families (between 6 and 7). In contrast, samples from 10 km downstream the cheese production factory presented higher biodiversity, many pollution-sensitive taxa such as Athericidae, Perlidae, Perlodidae and Sericostomatidae and a number of families between 26 and 27 [80].

Cheese whey wastewater has also shown to be toxic to aquatic organisms. Toxicological results of the zebrafish *Danio rerio* embryo bioassay with a mean 7-day LC_{50} was 0.655%, while bioassays on *Daphnia magna* and *Thamnocephalus platyurus* presented a higher LC_{50} value of 3.032% and 1.56%, respectively [80]. Even after treatment with an anaerobic fermentation system for hydrogen production, the CWW samples varied from "very" to "extremely toxic" [81]. Average toxicity values for the zebrafish *Danio rerio* embryo bioassay were 1.55% (24 h) and 0.75% (48 h), for *Thamnocephalus platyurus* 0.69% (24 h) and for *Daphnia magna* 2.51% (24 h) and 1.82% (48 h). Toxicity of CWW was attributed to the chemical compounds PO_4^{-3}, SO_4^{-2}, N-NH_3 and NO_3^- [81].

Similar to olive mills, cheese producing plants are small capacity units that are scattered throughout Greece and cause very serious environmental problems due to their large volume and organic load of wastewaters. To date, an integrated treatment solution at national level has not been implemented, despite the existence of various small-scale treatment technologies. For example, a new, integrated technology for the treatment and utilisation of cheese-dairy wastewater has been developed by the laboratory of Organic Chemical Technology of the National Technical University of Athens and has successfully been tested in a cheese-making factory in Viotia [74]. The proposed technology reduced fat and oil content by 76% and COD by 90%, while biogas was produced (4 m^3/h). The university laboratory concluded that the final effluent could be disposed in water bodies after an aerobic biological refining, however the final effluent should be tested for toxicity as the effluent can still be toxic after treatment as has been shown in some studies (e.g. [81]).

5 Other Agroindustrial Industries

Greece's running waters are also recipients of effluents from other agricultural industries, including animal factory farms, dairy farms, slaughterhouses, food, fruit and meat processing plants, tannery (leather processing plants), wineries and paper and weaving (cotton and textile) industries. The common characteristic of all these industries is the high organic content (BOD and COD) and total solids [82] of their wastewaters that result in oxygen depletion when discharged in receiving waterways [83, 84]. The acceptable lower limit for oxygen concentrations in rivers is usually about 6 mg/l, which is the level in which sensitive fish species (usually trout and salmon) are able to survive [85]. The discharge of organic wastewaters to rivers results in the development of bacterial or fungal-dominated epilithon (stone-attached communities), commonly referred to as sewage fungus or sewage bacteria [86]. These growths degrade river aesthetics, make the riverbed unsuitable for fish and many invertebrate species [19, 28, 44] and gradually decrease the water pH, accompanied by a release of strong odours due to decomposition of organic compounds. The receiving water becomes a breeding place for pollution-tolerant species, which are usually Dipteran species, such as flies and mosquitoes, and may often be carriers of dangerous diseases.

These effects have been observed in many river systems throughout Greece, but no prevention and control measures are implemented. Usually, pollution incidents become obvious when mass fish deaths are witnessed by the local inhabitants. A recent example comes from the Spercheios River in Central Greece, where hundreds of fish died due to oxygen depletion. Fish mortality is attributed to untreated wastewater discharge from the paper mill into a tributary of Spercheios (Asopos stream). The mill operated all year round, but effects are more pronounced during the dry period where flow is at a minimum. Ecological quality of the stream near the paper mill was classified as poor and bad, and only *Chironomus plumosus*-gr and species of the Simuliidae family were detected at very high abundances (Karaouzas et al., unpublished results). The effects of paper mill wastewaters into river ecosystems are documented elsewhere [87, 88].

Although the livestock industry is one of the major industries in Greece, its effects on the environment have been largely overlooked and ignored. The United Nations has declared concentrated animal feeding operations to be "one of the top two or three most significant contributors to the most serious environmental problems, at every scale from local to global" [100]. Wastewater discharge from slaughterhouses causes deoxygenation of rivers [84] and contamination of groundwater [89]. Blood, one of the major dissolved pollutants in slaughterhouse wastewater, has a chemical oxygen demand (COD) of 375 g/l and contains high concentrations of slowly biodegradable suspended solids, including pieces of fat, grease, hair, feathers, flesh, manure, grit, and undigested feed [90]. Furthermore, livestock produce significant amounts of manure, which may overflow due to heavy rainfalls or ruptures leach through the soil into groundwater [91, 92]. Manure is rich in compounds of nitrogen, phosphorus and ammonia. When excessive amounts of these compounds enter into freshwaters, they can lead to lethal algal blooms, causing eutrophication [92].

Elevated nutrient levels due to livestock and food processing industries have been recorded in several rivers of Greece. A general increasing trend of the annual mean values of nitrogen and phosphorous compounds at the Louros River, at its conjunction with the small tributary of Vossa which receives wastes from animal farms, has been observed [93]. Increased values of organic matter (8%) have been found at a site of Asmaki canal (Larissa, Thessaly), where a textile-dyeing plant is operating [94]. Two other sampling sites, in the same canal area where extensive farming and an alcohol producing factory occur, displayed high Cu values due to increased organic matter which strongly retains Cu [94].

High nutrient levels (NH_3-N, NO_2-N and NO_3-N) have also been recorded in Canal 66 that flows through Veria city and discharges into the Aliakmon River [66]. Canal 66 receives agroindustrial wastewaters mainly from canneries, and concentrations are higher during the low flow season than in the high flow season due to the lower discharge [66]. This canal is considered by many the most polluted freshwater body of Northern Greece, and in the press it is often cited as “The Canal of Death” due to the frequent sighting of mass fish deaths.

Concluding, it must be noted that all major rivers in Greece have significant pollution problems, particularly at their downstream parts due to wastewater discharge. These rivers receive pollutants from many other point and non-point pollution sources, thus further deteriorating their ecological quality.

6 Conclusions

Agro-industrial wastewater management is today one of the main concerns for ensuring a sustainable environment. Management of wastewaters, as covered in this chapter, is crucial in view of the high organic matter and high nutrient levels that they contain. Most of these wastewaters can be effectively treated either with aerobic or anaerobic digestion processes [74, 82, 95, 96]. Furthermore, all these wastewaters contain nutrients, salts, organics and oils that can be recycled or utilised for other purposes and with effective treatment can be used to irrigate pasture, thereby conserving potable water. Occasionally, pretreatment strategies (i.e. wetlands, artificial lagoons) are required in order to improve the efficiency of the treatment methodology. In the agricultural sector, methane recovery and use as a clean energy source can be a highly sustainable solution, contributing to a number of environmental objectives, as well as providing social and economic benefits for rural communities.

Finally, and most importantly, new regulations must be implemented for the treatment and management of these agroindustrial wastewaters. These wastewaters are usually discharged in small stream catchments (<10 km^2) which are not considered in the Water Framework Directive 2000/60/EC. Therefore, there is a need for including small streams into monitoring and assessment schemes as small streams contribute to the pollution load of the river basin. Furthermore, guidelines to manage these wastes through technologies that minimise their environmental impact and lead to a sustainable use of resources are critical.

Acknowledgements Many thanks to my colleagues Yannis Kapakos and Katerina Vourka for their help in literature search and data collection.

References

1. Arthington AH, Welcomme RL (1995) The condition of large river systems of the World. In: Armantrout NB, Wolotira RJ (eds) Condition of the world's aquatic habitats. World fisheries congress. Science, Boca Raton
2. Mason CF (1996) Biology of freshwater pollution, 3rd edn. Longman, London
3. Niaounakis M, Halvadakis CP (2006) Olive processing waste management – literature review and patent survey, 2nd edn. Elsevier, Amsterdam
4. FAO (2014) Food and agriculture organization of the United Nations. Statistics division. http://faostat3.fao.org/home/E. Accessed 26 Nov 2014
5. Kapellakis IE, Tsagarakis KP, Avramaki C, Angelakis AN (2006) Olive mill wastewater management in river basins: a case study in Greece. Agric Water Manag 82:354–370
6. Paraskeva P, Diamadopoulos E (2006) Technologies for olive mill wastewater (OMW) treatment: a review. J Chem Technol Biotechnol 81:1475–1485
7. Kavvadias V, Doula MK, Komnitsas K, Liakopoulou N (2010) Disposal of olive oil mill wastes in evaporation ponds: effects on soil properties. J Hazard Mater 182:144–155
8. Davies LC, Vilhena AM, Novias JM, Martins-Dias S (2004) Olive mill wastewater characteristics: modeling and statistical analysis. Grasas Aceites 55:233–241
9. Alburquerque JA, Gonzalvez J, Garcia D, Cegarra J (2004) Agrochemical characterisation of 'alperujo', a solid by-product of the two phase centrifugation method for olive oil extraction. Bioresour Technol 92:195–200
10. Vlyssides AG, Loizidou M, Gimouhopoulos K, Zorpas A (1998) Olive oil processing wastes production and their characteristics in relation to olive oil extraction methods. Fresenius Environ Bull 7:308–313
11. De Marco E, Savarese M, Paduano A, Sacchi R (2007) Characterization and fractionation of phenolic compounds extracted from olive oil mill wastewaters. Food Chem 104:858–867
12. Arienzo M, Capasso R (2000) Analysis of metal cations and inorganic anions in olive mill wastewaters by atomic absorption spectroscopy and ion chromatography. Detection of metals bound to the organic polymeric fraction. J Agric Food Chem 48:1405–1410
13. Hamdi M (1993) Future prospects and constraints of olive mill wastewaters use and treatment: a review. Bioprocess Biosyst Eng 8:209–214
14. Karaouzas I, Cotou E, Albanis TA, Kamarianos A, Skoulikidis NT, Giannakou U (2011) Bioassays and biochemical biomarkers for assessing olive mill and citrus processing wastewater toxicity. Environ Toxicol 24:296–303
15. Paixão SM, Mendoņca E, Picado A, Anselmo AM (1999) Acute toxicity evaluation of olive mill wastewaters: a comparative study of three aquatic organisms. Environ Toxicol 14:263–269
16. Roig A, Cayuela ML, Sanchez-Monedero MA (2006) An overview on olive mill wastes and their valorisation methods. Waste Manage 26:960–969
17. Fiorentino A, Gentili A, Isisori M, Monaco P, Nardelli A, Parrela A, Temussi F (2003) Environmental effects caused by olive mill wastewaters: toxicity comparison of low-molecular-weight phenol components. J Agric Food Chem 51:1005–1009
18. Mekki A, Dhouib A, Feki F, Sayadi S (2008) Assessment of toxicity of the untreated and treated olive mill wastewaters and soil irrigated by using microbiotests. Ecotoxicol Environ Saf 69:488–495
19. Karaouzas I, Skoulikidis NT, Giannakou U, Albanis TA (2011) Spatial and temporal effects of olive mill wastewaters to stream macroinvertebrates and aquatic ecosystems status. Water Res 45:6334–6346
20. Rouvalis A, Iliopoulou-Georgudaki J, Lyberatos G (2004) Application of two microbiotests for acute toxicity evaluation of olive mill wastewaters. Fresenius Environ Bull 13:458–464
21. Yesilada O, Sam M (1998) Toxic effects of biodegraded and detoxified olive oil mill wastewater on the growth of *Pseudomonas aeruginosa*. Environ Toxicol Chem 65:87–94

22. DellaGreca M, Monaco P, Pinto G, Pollio A, Previtera L, Temussi F (2001) Phytotoxicity of low-molecular-weight phenols from olive mill waste waters. Bull Environ Contam Toxicol 67:352–357
23. Kotsou M, Mari I, Lasaridi K, Chatzipavlidis I, Balis C, Kyriacou A (2004) The effect of olive oil mill wastewater (OMW) on soil microbial communities and suppressiveness against *Rhizoctonia solani*. Appl Soil Ecol 26:113–121
24. Rinaldi M, Rana G, Introna M (2003) Olive-mill wastewater spreading in southern Italy: effects on a durum wheat crop. Field Crop Res 84:319–326
25. Aggelis G, Iconomou D, Christou M, Bokas D, Kotzailias S, Christou G, Tsagou V, Papanikolaou S (2003) Phenolic removal in a model olive oil mill wastewater using *Pleurotus ostreatus* in bioreactor cultures and biological evaluation of the process. Water Res 37: 3897–3904
26. Venieri D, Rouvalis A, Iliopoulou-Georgudaki J (2010) Microbial and toxic evaluation of raw and treated olive oil mill wastewaters. J Chem Technol Biotechnol 85:1380–1388
27. Gotsi M, Kalogerakis N, Psillakis E, Samaras P, Mantzavinos D (2005) Electrochemical oxidation of olive oil mill wastewaters. Water Res 39:4177–4187
28. Karaouzas I (2011) Ecotoxicity and effects of agro-industrial wastewaters to stream biodiversity at multiple levels of biological organization. PhD Thesis. University of Ioannina
29. Persoone G, Marsalek B, Blinova I, Törökne A, Zarina D, Manusadzianas L, Nalecz-Jawecki G, Tofan L, Stepanova N, Tothova L, Kolar B (2003) A practical and user-friendly toxicity classification system with microbiotests for natural waters and wastewaters. Environ Toxicol 18:395–402
30. Charistou A, Kyriakopoulou K, Anastasiadou P, Fokialakis N, Skaltsounis AL, Machera K (2010) Acute toxicity of Olive Mill Wastewater on rats, *Vibrio fischeri* and *Artemia franciscana*. Hellenic Plant Protection J 3:17–24
31. Podimata M, Koutseris E, Tsiropoulos N (2004) Water quality assessment by ecotoxicological and chemical methods in Magnesia, Greece. Water Air Soil Pollut Focus 4:179–187
32. Rouvalis A, Iliopoulou-Georgudaki J (2010) Comparative assessment of olive oil mill effluents from three-phase and two-phase systems, treated for hydrogen production. Bull Environ Contam Toxicol 85:432–36
33. Anastasopoulou E, Pavlidou A, Dassenakis E, Hatzianestis I (2012) Determining the toxicity of olive oil mill waste waters using aquatic macroinvertebrates. Proceedings of the olive oil mill wastes and environmental protection symposium, Chania, 16–18 October 2012
34. Pavlidou A, Anastasopoulou E, Dassenakis M, Hatzianestis I, Paraskevopoulou I, Simboura N, Rousselaki E, Drakopoulou P (2014) Effects of olive oil wastes on river basins and an oligotrophic coastal marine ecosystem: a case study in Greece. Sci Total Environ 497–498:38–49
35. Danellakis D, Ntaikou I, Kornaros M, Dailianis S (2011) Olive oil mill wastewater toxicity in the marine environment: alterations of stress indices in tissues of Mussel Mytilus galloprovincialis. Aquat Toxicol 101:358–366
36. Ntougias S, Gaitis F, Katsaris P, Skoulika S, Iliopoulos N, Zervakis GI (2013) The effects of olives harvest period and production year on olive mill wastewater properties – evaluation of Pleurotus strains as bioindicators of the effluent's toxicity. Chemosphere 92(4):399–405
37. Ginos A, Manios T, Mantzavinos D (2006) Treatment of olive mill effluents by coagulation-flocculation-hydrogen peroxide oxidation and effect on phytotoxicity. J Hazard Mater 133: 135–142
38. Ouzounidou G, Asfi M, Sotirakis N, Papadopoulou P, Gaitis F (2008) Olive mill wastewater triggered changes in physiology and nutritional quality of tomato (*Lycopersicon esculentum* mill) depending on growth substrate. J Hazard Mater 158:523–530
39. Asfi M, Ouzounidou G, Moustakas M (2012) Evaluation of olive oil mill wastewater toxicity on spinach. Environ Sci Pollut Res Int 19:2363–2371
40. Dalis D (1989) Anaerobic biological treatment of olive mill wastewater for biogas production. Management of olive mill wastewaters meeting, Heraklion, Crete

41. Anastasopoulou E, Skoulikidis N, Pavlidou A, Dassenakis E (2013) Effects of untreated olive oil waste water disposals in river epis and its estuaries in Messinia, Greece. In: Proceedings of the 4th international cemepe and secotox conference, Mykonos Island, 24–28 June 2013
42. Karaouzas I, Lambropoulou DA, Skoulikidis NT, Albanis TA (2011) Levels, sources and spatiotemporal variation of nutrients and micropollutants in small streams of a Mediterranean River basin. J Environ Monit 13:3064–3074
43. Tzoraki O, Moraetis D, Stamati F, Navrozidis S, Kouseri E, Kalogerakis N, Nikolaidis NP (2008) Fate of olive mill wastewater in evrotas river basin. In: Proceedings of the 1st international conference hazardous waste management, Chania, Crete
44. Voreadou K (1993) Effect of olive mill wastewaters on ecosystems of running waters of Crete. PhD Thesis. University of Crete
45. Voreadou K (1994) Liquid wastes from olive mills: effects on the natural water ecosystems of Crete – present management of the wastes and tendencies. In: Proceedings of the conference of management of olive wastes, Siteia, Crete
46. Skoulikidis N, Amaxidis Y, Bertahas I, Laschou S, Gritzalis K (2006) Analysis of factors driving stream water composition and synthesis of management tools – A case study on small/medium Greek catchments. Sci Total Environ 362:205–241
47. Anastasopoulou E, Georgiopoulos N, Katsogiannou I, Karaouzas I, Gritzalis K (2014) Olive oil waste waters and their impact on the benthic diversity of river Epis in Messinia, Greece. In: Proceedings of the 36th scientific conference of hellenic association for biological sciences Ioannina, 8–10 May 2014
48. Gritzalis K (2014) Monitoring of the quality and assessment of the ecological status of the Messinian rivers in South Peloponnese, Greece, Pamisos, Aris, Ligdou, Epis, Karias, Tzanes-Polylimnio, Maurozoumena, Despotis, Mourtia, Arkadikos, Nedas and Velikas. Final technical report. Hellenic Centre for Marine Research, Anavissos [in Greek]
49. Buffagni A, Erba S, Furse MT (2007) A simple procedure to harmonize class boundaries of assessment systems at the pan-European scale. Environ Sci Policy 10(7–8):709–724
50. Armitage PD, Moss D, Wright JF, Furse MT (1983) The performance of a new biological water quality score system based on macroinvertebrates over a wide range of unpolluted running water sites. Water Res 17:333–347
51. Doula M, Kavvadias V, Theocharopoulos S, Kouloumbis P, Oikonomou D, Arapoglou D (2009) Environmental impacts relative to soil quality caused from the disposal of olive oil mills wastes. Case study: A municipality in Crete, Greece. In: Proceedings of the 3rd AMIREG international conference: assessing the footprint of resource utilization and hazardous waste management, 2009, Athens, 84–89
52. Spandre R, Dellomonaco G (1996) Polyphenols pollution by olive mill waste waters, Tuscany, Italy. J Environ Hydrol 4:1–13
53. Abad-García B, Berrueta LA, Garmón-Lobato S, Urkaregi A, Gallo B, Vicente F (2012) Chemometric characterization of fruit juices from Spanish cultivars according to their phenolic compound contents: I. Citrus fruits. J Agric Food Chem 60:3635–3644
54. Goodrich RM, Braddock RJ (2006) Major by-products of the Florida citrus processing industry. IFAS, University of Florida: US. Publication Number FS10700
55. Koo RCJ (1973) Irrigation of citrus with citrus processing waste water. In: Proceedings of the Florida State Horticultural Society, vol 86, pp 233–237, Miami, Florida
56. Parish ME, Braddock RJ, Graumlich TR (1986) Chemical and microbial characterization of citrus oil-mill effluent. J Food Sci 51:431–433
57. Burton SG, Garcin CR and Aucamp JH (2007) Beneficiation of wastewaters from the South African citrus industry – a feasibility study. WRC Report No KV187/07. ISBN No 978-1-77005-562-2.
58. Di Mauro A, Fallico B, Passerini A, Maccarone E (2000) Waste water from citrus processing as a source of hesperidin by concentration on styrene-divinylbenzene resin. J Agric Food Chem 48:2291–2295

59. Hawash S, Hafez AZ, El-Diwani G (1988) Citrus processing wastewater treatment. Chem Ing Tech 60:128–130
60. Zafiris C, Boukis I, Choudalis P, Chatzithanasiou A, Holm–Nielsen JB, Baadstorp L, Karamanlis X, Kamarianos A, Kyriakis S, Zournas M (2000) Greek biogas production from pig manure and co-digestions. In: Renewable energy for Europe – campaign for take-off: Altener 2000 conference, pp 407–410, Toulouse, France, 23–25 October 2000
61. Bellos D (2004) The concentration of nutrients, heavy metals and radioactive in water, sediment and water plants in river Pinios. PhD Thesis, Aristotle University of Thessaloniki [in Greek]
62. Chatzinikolaou Y, Dakos V, Lazaridou M (2006) Longitudinal impacts of anthropogenic pressures on benthic macroinvertebrate assemblages in a large transboundary Mediterranean river during the low flow period. Acta Hydrochim Hydrobiol 34:453–463
63. Dassenakis M, Scoullos M, Foufa E, Krasakopoulou E, Pavlidou A, Kloukiniotou M (1997) Effects of multiple source pollution on a small Mediterranean river. Appl Geochem 13: 197–211
64. Iliopoulou-Georgudaki J, Kantzaris V, Katharios P, Kaspiris P, Georgiadis T, Montesantou B (2003) An application of different bioindicators for assessing water quality: a case study in the rivers Alfeios and Pineios (Peloponnisos Greece). Ecol Indic 2:345–360
65. Karaouzas I, Gritzalis KC, Skoulikidis NT (2007) Land use effects on macroinvertebrate assemblages and stream quality along an agricultural river basin. Fresenius Environ Bull 16: 645–653
66. Lazaridou-Dimitriadou M, Artemiadou V, Yfantis G, Mourelatos S, Mylopoulos Y (2000) Contribution to the ecological quality of Aliakmon river (Macedonia, Greece): a multivariate approach. Hydrobiologia 410:47–58
67. Lekkas T, Kolokythas G, Nikolaou A, Kostopoulou M, Kotrikla A, Gatidou G et al (2004) Evaluation of the pollution of the surface waters of Greece from the priority compounds of List II, 76/464/EEC Directive and other toxic compounds. Environ Int 30:995–1007
68. Theodoropoulos C, Iliopoulou-Georgudaki J (2010) Response of biota to land use changes and water quality degradation in two medium-sized river basins in southwestern Greece. Ecol Indic 10:1231–1238
69. Demirel B, Yenigun O, Onay TT (2005) Anaerobic treatment of dairy wastewaters: a review. Process Biochem 40:2583–2595
70. Watkins M, Nash D (2010) Dairy factory wastewaters, their use on land and possible environmental impacts – a mini review. TOASJ 4:1–9
71. Carvalho F, Prazeres AR, Rivas J (2013) Cheese whey wastewater: characterization and treatment. Sci Total Environ 445–446:385–396
72. Gavala HN, Kopsinis H, Skiadas IV, Stamatelatou K, Lyberatos G (1999) Treatment of dairy wastewater using an upflow anaerobic sludge blanket reactor. J Agric Eng Res 73:59–63
73. Rivas J, Prazeres AR, Carvalho F, Beltrán F (2010) Treatment of cheese whey wastewater: combined coagulation–flocculation and aerobic biodegradation. J Agric Food Chem 58: 7871–7877
74. Vlyssides A, Barampouti EM, Mai S, Tsimas E (2009) Anaerobic digestion of chemically pretreated wastewater from a cheese making factory. In: Proceedings of the 11th international conference on environmental science and technology, Crete
75. Vlyssides A, Tsimas E, Barampouti EM, Mai S (2012) Anaerobic digestion of cheese dairy wastewater following chemical oxidation. Biosyst Eng 113:253–258
76. Fritzscha B (1986) Soil geological and hydrogeological problem and the disposal of dairy product waste waters. Z Angew Geol 32:131–134
77. McAuliffe KW, Scotter DR, Macgregor AN, Earl KW (1982) Casein whey waste water effects on soil permeability. J Environ Qual 11:31–34
78. Marwaha SS, Kennedy JF (1988) Review: whey-pollution problem and potential utilization. Int J Food Sci Technol 23:323–336

79. Yang SY, Jones JH, Olsen FJ, Peterson JJ (1980) Soil as a medium for dairy liquid waste disposal. J Environ Qual 9:370–372
80. Karadima A, Theodoropoulos C, Rouvalis A, Iliopoulou-Georgudaki J (2010) Ecological risk assessment of cheese whey effluents along a medium-sized river in Southwest Greece. J Environ Sci Health A 45:775–781
81. Karadima C, Theodoropoulos C, Iliopoulou-Georgudaki J (2009) Environmental hazard assessment of cheese manufacturing effluent treated for hydrogen production. Bull Environ Contam Toxicol 83:428–434
82. Albanis T (2005) Pollution and environmental protection technologies, 3rd edn. University of Ioannina, Epirus
83. Hynes HBN (1960) The biology of polluted waters. Liverpool University Press, Liverpool
84. Quinn JM, McFarlane PN (1989) Effects of slaughterhouse and dairy factory wastewaters on epilithon: a comparison in laboratory streams. Water Res 23:1267–1273
85. Kemker C (2013) Dissolved oxygen. Fundamentals of environmental measurements. Fondriest environmental. http://www.fondriest.com/environmental-measurements/parameters/waterquality/dissolved-oxygen. Assessed 19 Nov 2013
86. Curtis EJC (1969) Sewage fungus: its nature and effects. Wat Res 3:289–311
87. Galloway BJ, Munkittrick KR, Currie S, Gray MA, Curry RA, Wood CS (2003) Examination of the responses of slimy sculpin (*Cottus cognatus*) and white sucker (*Catostomus commersoni*) collected on the Saint John River (Canada) downstream of pulp mill, paper mill, and sewage discharges. Environ Toxicol Chem 22:2898–2907
88. Karrasch B, Parra O, Cid H, Mehrens M, Pacheco P, Urrutia R, Valdovinos C, Zaror C (2006) Effects of pulp and paper mill effluents on the microplankton and microbial self-purification capabilities of the Biobýo River, Chile. Sci Total Environ 359:194–208
89. Sangodoyin AY, Agbawhe OM (1992) Environmental study on surface and groundwater pollutants from abattoir effluents. Bioresour Technol 41:193–200
90. Bull MA, Sterritt RM, Lester JN (1982) The treatment of wastewaters from the meat industry: a review. Environ Technol Lett 3:117–126
91. Burkholder J, Libra B, Weyer P, Heathcote S, Kolpin D, Thorne PS, Wichman M (2007) Impact of waste from concentrated animal feeding operation on water quality. Environ Health Perspect 115:308–312
92. Mallin MA, Cahoon LB (2003) Industrialized animal production – a major source of nutrient and microbial pollution to aquatic ecosystems. Popul Environ 24:369–385
93. Ovezikoglou V, Ladakis M, Dassenakis M, Scoullos M (2003) Nitrogen, phosphorus and organic carbon in main rivers of the Western Greece. Global Nest J 5:147–56
94. Augoustis A, Hatziioannou M, Papadopoulos S, Kateris D, Neofytou C, Vafidis D (2012) Assessing quality of an irrigation canal ecosystem, through water and sediment environmental parameters. A case study in Thessaly region, Greece. Wseas Trans Environ Develop 8(1): 23–32
95. Arvanitoyannis IS, Giakoundis A (2006) Current strategies for dairy waste management: a review. Crit Rev Food Sci Nutr 46:379–390
96. Justino C, Pereira R, Freitas A, Rocha-Santos T, Panteleitchouk T, Duarte A (2012) Olive oil mill wastewaters before and after treatment: a critical review from the ecotoxicological point of view. Ecotoxicology 21:615–29
97. Ministry of Rural Development and Food (2014) Statistical services. http://www.minagric.gr/index.php/el/xrisimewplirofories-2/statistika-politi. Assessed 14 Nov 2014
98. Savvides G (1994) Olive mill wastewaters and aquaculture. In: Proceedings of the international conference on olive mill waste Management. GEOTEE – Department of Crete, Sitia Development Organization, Sitia, June 16–17, pp 42–46
99. ELOGAK (2015) Hellenic Milk and Meat Agency. Statistics and data. http://www.elogak.gr. Accessed 22 April 2015
100. Steinfeld H, Gerber P, Wassenaar T, Castel V, Rosales M, de Haan C (2006) Livestock's long shadow: environmental issues and options. FAO, Rome. ftp://ftp.fao.org/docrep/fao/010/a0701e/a0701e00.pdf

Overview of the Pesticide Residues in Greek Rivers: Occurrence and Environmental Risk Assessment

Dimitra Lambropoulou, Dimitra Hela, Anastasia Koltsakidou, and Ioannis Konstantinou

Abstract During the past decades, there has been a growing concern related to the presence of "emerging" and "priority substances" in surface waters. For this reason, the European Union (EU) Water Framework Directive (WFD) (Directive 2000/60/EC) has established the bases to regulate water resources with the objective of preserving, protecting and improving their quality and sustainable use. In this context, this chapter provides an overview on the occurrence and levels of pesticide residues in Greek river waters, over the last 30 years [between 1985 and 2015; "past" (1985–2005) and "recent" (2006–2015) pesticide investigations] in order to describe trends in water quality status and assess potential adverse effects in the aquatic environment. The assembled data clearly demonstrate that agricultural practices in Greece have aggravated the water quality of rivers and may have posed considerable risk for certain ecosystems. The rivers that were monitored in a systematic way, mainly in the past period, were Aliakmon, Axios, Loudias, Louros, Arachthos and Kalamas, while there is a lack of data for other important rivers. Most of the detections involve a limited number of herbicides used extensively in corn, cotton and rice production, as well as the banned organochlorine insecticides that are persistent in the aquatic environment. Overall, the concentrations detected throughout the running waters of Greece were very small fractions of levels that, according to environmental risk assessment analysis, are believed to be non-harmful to aquatic life. However, in some areas with intense agricultural

D. Lambropoulou (✉) and A. Koltsakidou
Department of Chemistry, Aristotle University of Thessaloniki, Thessaloniki, Greece
e-mail: dlambro@chem.auth.gr

D. Hela
Department of Business Administration of Food and Agricultural Enterprises, University of Patras, Agrinio 30100, Greece

I. Konstantinou
Department of Environmental and Natural Resources Management, University of Patras, Agrinio 30100, Greece

N. Skoulikidis et al. (eds.), *The Rivers of Greece: Evolution, Current Status and Perspectives*, Hdb Env Chem (2018) 59: 205–240, DOI 10.1007/698_2015_428,

practices and hence high pesticide application, the environmental concentrations of pesticides were in non-compliance with the environmental quality standards (EQSs, Directive 2008/105/EC). The outcomes of this review reveal that there is a need for harmonisation in the sampling strategy and monitoring practices should be in accordance with the WFD. For future campaigns, specific insight into agricultural treatments and land use in the river basin could contribute to optimised water monitoring.

Keywords Environmental risk assessment, Greek rivers, Occurrence, Pesticides

Contents

1 Introduction

Pesticides have been widely used throughout the world since the middle of the last century. Around 1,000 active ingredients have been employed and are currently formulated in thousands of different commercial products. It has been estimated that, in 2001, 2.26 million metric tons of such substances were applied all over the world. They include a variety of compounds, mainly insecticides, herbicides and fungicides, with varying physicochemical characteristics such as polarity, volatility, persistence, etc. Many public health benefits have been obtained from the use of synthetic pesticides, and their use in agriculture has been one of the most important factors leading to increased yields and reduced product prices.

In spite of these obvious advantages, the detection of pesticide residues in various sections of the environment has raised serious concerns regarding their use. The publication of Rachel Carson's *Silent Spring*, which highlighted the risks of pesticide use, stimulated the steady progress in documenting the negative spillovers arising from the excessive, uncontrolled and continuous use of chemical inputs (e.g. toxic effects on humans, livestock and wildlife, adverse effects on target biota, pest resistance, etc.) [1, 2]. In response to these concerns, a plethora of studies have been published during the last decades covering different topics related to pesticide pollution sources and processes, occurrence, fate and potential toxicity [3, 4]. In addition, numerous local and national monitoring strategies have been undertaken in many countries in order to quantify the amount of pesticides entering the environment and to monitor ambient levels, trends and potential effects.

Despite, however, the vast research performed around the world providing a nationwide pattern on pesticide occurrence and distribution, a relative small number of studies have been conducted in Greece. In general, research and monitoring data on the environmental occurrence of pesticides in the surface waters of Greece have been limited to studies focusing on a small number of targeted compounds in localised areas, mainly in Northern Greece. For example, only limited retrospective monitoring data are available in all water compartments (rivers, lakes, groundwater, etc.), and a lack of monitoring data is observed for many pesticides both in space and time. Therefore, the present chapter intends to provide a comprehensive overview of pesticide occurrence in Greek rivers. In particular, data published over the last 30 years (from 1985 to 2015) were reviewed with the scope: (a) to identify which agrochemicals are important to the contamination of the aquatic environment taking into account their environmental concentrations and toxicological properties, (b) to identify the possible relative risks posed to aquatic life, (c) to evaluate the compliance of the detected pesticide concentrations with environmental quality standards (EQSs) of the Water Framework Directive (WFD) and (d) to highlight the trends on the concentration levels of "old" and "modern" pesticides between different decades, over the last 30 years. Therefore, data were divided into two groups – first group (1985–2005) and second group (2006–2015) – and were treated and discussed in detail as "past" and "recent" pesticide investigations, respectively.

2 Fate and Pathways of Pesticides in the Aquatic Environment

Pesticide transport and transformation are the two main routes that affect pesticide availability and efficacy. Pesticides may be transferred to water bodies through point and non-point sources. Point source can be any single identifiable source of pollution from which pesticides are discharged such as the effluent pipe from pesticide factories, careless storage and handling (e.g. disposal of pesticide containers), accidental spills and overspray. Non-point source pollution is defined as the pesticide movement away from the targeted application site (often referred to as "off-target" site). Because of its diffuse nature, non-point source pollution typically yields relatively uniform environmental concentrations of pesticides in surface waters, sediments and groundwater [5]. Among the non-point sources, surface run-off and erosion, spray drift, leaching and drain flows (preferential flow) are probably the most important. Pesticide diffuse pollution can also occur through atmospheric transport by volatilisation and subsequent deposition.

Next to pesticide transfer, pesticide transformation can also play an important role in their environmental behaviour in the aquatic environment. In general, chemical, physical and biological factors can be responsible for pesticide transformation [5–7]. Under field conditions, breakdown of pesticides (photolysis, chemical and microbial breakdown) is influenced by a combination of these factors, such as temperature, pH, ionic strength, light, the presence of suspended solids and dissolved organic matter, microbial activity, etc. [5–7].

In addition, environmental factors such as soil characteristics, topography, weather as well as agricultural and application practices can also influence pesticide levels and their persistence in the aquatic environment [5–7]. Hence, their environmental fate can be estimated based on physicochemical parameters, including water solubility, the octanol–water partition coefficient (K_{ow}), dissociation constant (K_a), soil sorption coefficient (K_{oc}), vapour pressure and bioconcentration factor (BCF) [8]. For example, pesticides, which are sufficiently resistant to degradation and are adequately soluble to be transported in water, may reach the rivers in significantly greater amounts than other pesticides that presented fast degradation and increased adsorption on soil.

3 Pesticide Use in Europe and Greece

In 2011, the market for crop protection products in Europe [EU-27 and European Free Trade Association (EFTA) nations] increased by 7.2% to reach €7,683 million at the manufacturer level [9] (Fig. 1). Market share of fungicides by volume is the highest, while insecticide use over the years has decreased. Levels of usage vary between countries with France, Spain, Italy and Germany being, by far, the largest markets in the EU-27 accounting for 61.903, 39.043, 37.630 and 31.425 tons of all pesticide ingredients, respectively (Fig. 2).

Pesticide usage in Greece is closely related to cropping patterns and is also subject to seasonal variation in response to climatological conditions. The overall amount of pesticides (including insecticide, fungicide, herbicide and other plant protection products) used during the last decade in agriculture varies between 14.921 (2007) and 6.537 (2009) tons per year (Fig. 3). On a weight basis, fungicides were used in the largest quantities, followed by herbicides together with insecticides. The average consumption of pesticides per hectare of treated area (including permanent arable land, forestry and foraging areas) was 2.8 kg/ha in 2000. The amount of pesticide used in terms of active ingredient has fallen since 2007, both in absolute amounts and in application rates. This might be partly explained by the new, more robust policies set up by the EU for the sustainable use of pesticides including the integrated pest management (IPM), the pesticide reduction practices and the introduction of low-dose pesticides. Of course, the financial crisis can also be considered as a major driving force for this reduction. Due to the economic

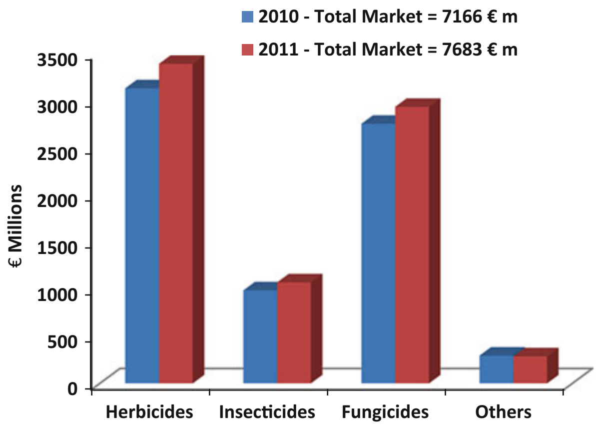


Fig. 1 Sales of pesticides (€ millions (m)) in the European market (EU-27 + EFTA) by product sector for 2011

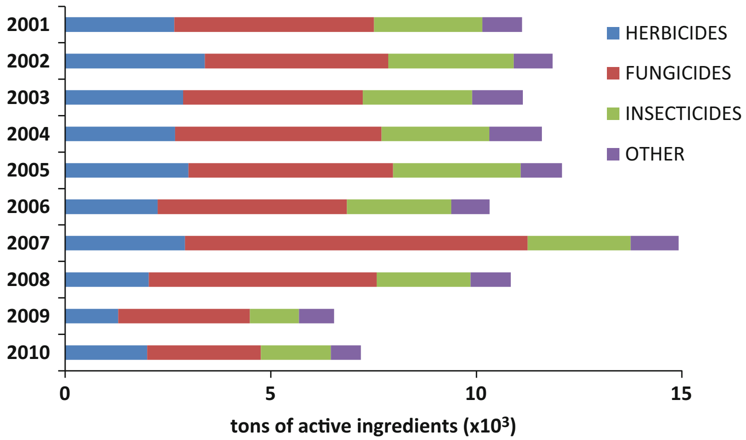


Fig. 2 Tons of active ingredients of pesticides used in Greece during the period 2001–2010

crisis, working conditions have dramatically deteriorated and farmers have been forced to reduce the use of pesticides and moved away from old pesticide practices (i.e. use of excessive dose/rates, preharvest intervals, illegal uses etc.).

4 Regulatory Framework and Water Framework Directive (WFD) for Pesticide Control

The potential adverse consequences derived from the use of pesticides have led to the development of special regulations by the European Commission (EC). Pesticide policies were first introduced at European Union (EU) level in 1979. The

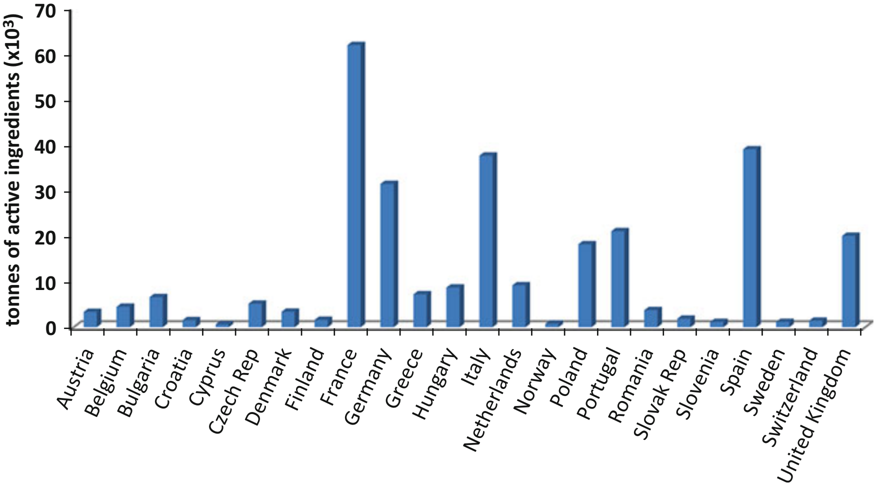


Fig. 3 Tons of active ingredients of pesticides used in Europe in 2010 (for the United Kingdom, data was available for the year 2009)

Directives 91/414/EC [10] and 98/8/EC [11] on the placing of plant protection products and biocidal products on the market were the first ones dealing with the authorisation of pesticides. Since then, many attempts have been made to evaluate pesticide risks and a number of other directives and regulations have also been introduced under the EU's pesticide policy framework. For instance, the Waste Framework Directive (2006/12/EC) [12] and the Directive on Hazardous Waste (91/689/EEC) [13] constitute regulations impacting pesticide pollution in many ways, as they establish provisions for the safe collection/disposal of empty pesticide packages and unused or expired pesticides. The specific actions for the sustainable use of pesticides have been refined in Directive 2009/128/EC [14] of the European Commission (EC). This directive establishes a framework to achieve a sustainable use of pesticides by reducing the risks and impacts of pesticide use on human health and the environment, while promoting the use of integrated pest management and of alternative approaches or techniques, such as non-chemical alternatives to pesticides. Regarding the quality of water intended for human consumption, Directive 98/38/EC sets a limit of 0.1 μg/L for a single active ingredient of pesticides and 0.5 μg/L for the sum of all individual active ingredients detected and quantified through monitoring, regardless of hazard or risk [15]. More recently, Directive 2006/118/EC on the protection of groundwater against pollution and deterioration sets a maximum of 0.1 μg/L for individual pesticides and 0.5 μg/L for the total pesticides (including active substances and their relative metabolites and transformation products) [16].

The Water Framework Directive (WFD, 2000/60/EC) [17], which describes the monitoring of priority substances and other pollutants in EU's surface waters, is widely recognised as one of the most important European environmental directive in the area of the pesticide control. The directive aims to achieve and ensure "good quality" status of all water bodies throughout Europe by 2015, and this is to be achieved by implementing management plans at the river basin level. It prefigures that water quality should be monitored on a systematic and comparable basis, and thus, technical specifications should follow a common approach (e.g. standardisation of monitoring, sampling and methods of analysis). Regarding the characterisation of the chemical status of surface water bodies, chemical monitoring is expected to intensify, following a list of 33 "priority chemicals" (inorganic and organic pollutants, one third of which are pesticides) (Decision 2455/2001 EC) [18], which will be reviewed every 4 years. The hazardous nature of "priority pollutants" is caused by their toxicity in combination with high chemical and biological stability and/or a high lipophilicity.

The successor Directive 2008/105/EU [19] of the European Parliament and the Council of the European Union has defined EQS values, i.e. annual averages (AA-EQS) and maximum allowable concentrations (MAC-EQS) for the priority substances in surface waters, with the aim to protect the aquatic environment from adverse effects of these substances. The concentrations of the priority substances in water, sediment or biota must be below the environmental quality standards (EQSs): this is expressed as "compliance checking". Compliance with AA-EQSs and MAC-EQSs defines the chemical status of the water body as "good". Under the WFD, member states must set quality standards (according to Annex V, 1.2.6) for "river basin-specific pollutants" (listed in Annex VIII, 1–9) that are "discharged in significant quantities" and take action to meet those quality standards by 2015 as part of the ecological status (Articles 4, 11, and Annex V, 1.3, WFD). EQSs are, therefore, key tools in assessing and classifying both chemical and ecological status. Among the priority substances, specific compounds have been classified as priority hazardous substances, with the aim to cease or phase out their discharges, emissions and losses.

The list of priority substances was recently revised (Directive 2013/39/EU) [20] and the number of priority substances was increased up to 45 (Table 1). New priority substances were added by the European Commission; among these are the biocides cybutryne (Irgarol) and terbutryn and the pesticides aclonifen, bifenox, cypermethrin and heptachlor/heptachlor epoxide. Irgarol and terbutryn are *s*-triazine compounds primarily used as algaecides/biocides in buildings, while Irgarol has also been used as an antifouling agent on ships, replacing the phased-out tributyltin (TBT). Aclonifen, bifenox and cypermethrin are pesticides in current use. Heptachlor is a chlorinated insecticide no longer used, which is mainly degraded to heptachlor epoxide in the environment. Production and use of heptachlor are regulated globally through the Stockholm Convention (SC) on Persistent Organic Pollutants (POPs) [21].

Table 1 Physicochemical properties and summary of environmental quality standards (EQSs) for annual averages (AA-EQSs) and maximum allowable concentrations (MAC-EQSs) of priority pesticides

	AA-EQS: inland surface water (μg/L)[a]	MAC-EQS: inland surface water (μg/L)[a]	S_w (mg/L)[b]	Log [b]K_{OW}	H (Pa m^3/mole)[b]	DT_{50}s (days)[b]	K_{oc} (mL/g)[b]
Aclonifen	0.120	0.12	1.40	4.37	3.03×10^{-3}	117	7,126[c]
Alachlor	0.300	0.70	240	3.09	3.20×10^{-3}	14	335
Atrazine	0.600	2.00	35	2.70	1.50×10^{-4}	75	100
Bifenox	0.012	0.04	0.10	3.64	1.62×10^{-4}	6	10,000
Chlorfenvinphos	0.100	0.30	145.00	3.80	1.55×10^{-3}	40	680.0
Chlorpyrifos	0.030	0.10	1.05	4.70	0.478	50	8,151
Cyclodiene pesticides	$\Sigma = 0.01$	Not applicable					
Aldrin			0.027	6.50	1.72×10^{1}	365	17
Dieldrin			0.23	3.70	6.50×10^{-2}	1,400	12
Endrin			0.24	3.20	1.48×10^{-1}	4,300	10
Isodrin			0.014	6.75	39.21	–	11
Cybutryne	0.003	0.016	7.00	3.95	–	–	1,569
Cypermethrin	8×10^{-5}	6×10^{-4}	0.009	5.3	2.00×10^{-2}	60	156,250
DDT total	0.025	Not applicable	0.006	6.91	8.43×10^{-1}	2,000	151,000
para-para-DDT	0.010	Not applicable	0.006	6.91	8.43×10^{-2}	2,000	151,000
Dichlorvos	6×10^{-4}	7×10^{-4}	18,000	1.90	2.58×10^{-2}	2	50
Dicofol	1.3×10^{-3}	Not applicable	0.8	4.30	2.45×10^{-2}	80	6,064
Diuron	0.200	1.80	35.6	2.87	2.00×10^{-6}	75.5	813
Endosulfan	0.005	0.01	0.32	4.75	1.48	50	11,500
Hexachlorobenzene	0.010	0.05	0.0047	5.73	35[d]	2,000	50,000
Hexachlorocyclohexane (lindane)	0.020	0.04	8.52	3.50	1.483×10^{-6}	980	1,270
Heptachlor	2×10^{-7}	3×10^{-4}	0.056	5.44	3.53×10^{2}	285	24,000
Heptachlor epoxide[e]	2×10^{-7}	3×10^{-4}	0.275	4.90	3.2	–	20,893

Isoproturon	0.3	1.00	70.2	2.50	1.46×10^{-5}	12	155
Simazine	1.000	4.00	5	2.30	5.60×10^{-5}	60	130
Trifluralin	0.030	Not applicable	0.221	5.27	10.2	181	15,800
Terbutryn	0.065	0.34	25	3.66	1.50×10^{-3}	74	2,432
Quinoxyfen	0.150	2.70	0.047	4.66	3.19×10^{-2}	97	22,929

AA annual average, *MAC* maximum allowable concentration, inland surface waters encompass rivers and lakes and related artificial or heavily modified water bodies, S_W solubility in water at 20°C, K_{OW} octanol–water partitioning coefficient at pH 7, 20°C, DT_{50s} soil half-life at 20–25°C, K_{oc} soil organic carbon–water partitioning coefficient

– Data not found

[a]Directive 2008/105/EC and 2013/39/EU of the European Parliament and of the European Council

[b]http://sitem.herts.ac.uk/aeru/iupac/index.htm

[c]http://www.pesticideinfo.org/

[d]Liisa M. Jantunen, Terry F. Bidleman,2006

[e]http://www.efsa.europa.eu

5 Analytical Methodologies in Pesticide Monitoring Programmes

The most widely used analytical methodologies for the analysis of pesticides in environmental waters are based on solid phase extraction (SPE) [22]. Extraction of pesticides from water samples is mostly performed by offline or online SPE by passing a water sample volume of 0.1–1 L through SPE cartridges or discs. C_{18}-bonded silicas, polymeric HLB OASIS and SDB are the sorbents more widely used. Method development in SPE is usually accomplished by investigating pH, sample matrix, polarity and flow rate of the sample and elution solvent and physicochemical characteristics of the sorbent bed [23]. Sample pH can be critical in order to obtain high yields of analyte retention in the sorbent. Consequently, in some cases, modification of sample pH can be necessary in order to stabilise the analytes, decrease the biological activity of the sample matrix and increase their retention in the solid phase. Although most official methods for pesticide analysis in water samples use liquid–liquid extraction (LLE) on account of its simplicity and consolidated status, this technique is not preferred for water monitoring studies [24]. Other methodologies, such as solid phase microextraction, have also been used to determine pesticides in waters [25].

In the past decades, the methods of trace-level determination of pesticides have changed considerably. For almost all studies performed between 1985 and 2005, the pesticide analysis was conducted using gas chromatography (GC) in combination with selective detectors such as flame-thermoionisation detector (FTD), nitrogen–phosphorus detector (NPD) and electron-capture detector (ECD) [26]. Despite their high sensitivity, the above-mentioned detectors offer only limited specificity and their use does not provide unambiguous identification and confirmation of a substance in real samples. Therefore, confirmation of results was also performed by using a gas chromatograph equipped with a different type of column or detector and hyphenated techniques such as gas chromatography–mass spectrometry (GC–MS), which is the most commonly used approach in the last decade.

Although GC has been proven successful for the analysis of nonpolar, semipolar, volatile and semi-volatile pesticides in environmental samples, for polar, non-volatile and thermally unstable pesticides, such as phenylureas, carbamates, pyrimidines, triazoles and phenoxyalkanoic acids, as well as for a large majority of all pesticide transformation products, GC is impossible or problematic and in this case liquid chromatography (LC) is the technique of choice.

The EU has introduced new definitions and criteria for confirmatory analysis, where chromatographic separation coupled to MS plays the leading role. According to 2002/657/EC [27] and EU SANCO guidelines [28], an unambiguous determination is based on a system of identification points (IPs) to score the MS data, in which the number of IPs given in MS analyses depends on the general degree of selectivity of the MS technique used. The more selective and specific the technique, the more information points are readily accumulated.

Although in the last few years LC–MS and LC–MS/MS have gained in popularity and are preferred for environmental water analysis of these compounds based on high sensitivity and selectivity, up until now, Greek pesticide studies in water samples are exclusively performed by the use of GC–MS [26, 29, 30]. However, this trend is expected to overturn fairly quickly and developments in the rapidly evolving area of LC–MS/MS for environmental pesticide analysis are anticipated shortly.

6 Overview of Geographical Distribution of the Research Studies in Greece

This study reviews articles that have been published between 1985 and 2015 and reported concentrations for a number pesticides (including metabolites and transformation products) in water samples from Greek rivers (Table 2, Fig. 4). Maximum concentration levels of detected pesticides in different Greek rivers during the sampling period of 1985–2005 are depicted in Figs. 5, 6, 7 and 8.

Table 2 Length, catchment area and cultivations of the main Greek rivers monitored for pesticide residues

	CA (km^2)	L (km)	Cultivation	References
Evros	53,078	550	Winter cereals	[30]
Strymonas	17,087	410	Wheat, maize ,tobacco, rice, corn, sunflower, sugar beets, vegetables	[31]
Axios	24,604	380	Cotton, corn, rice, fruit, tobacco and horticulture	[32]
Erythropotamos	7,982	350	Cotton, corn, sunflower, sugar beets	[30]
Aliakmon	8,880	310	Cotton, corn, rice, fruit trees (mainly peach and apples), sugar beet, vegetables and rare crops (alfalfa and grains)	[33, 34]
Ardas	5,795	290	Corn, sugar beets and sunflower	[30]
Pinios	10,743	257	Wheat, cotton, barley	[35]
Acheloos	6,478	255	Corn, olive trees, tobacco, cereals and vegetables	[29]
Nestos	6,265	246	Tobacco, sunflower, corn, vegetables	[36]
Arachthos	1,907	105	Citrus fruits, olives, corn, alfalfa and cotton	[37]
Kalamas	1,831	96	Maize, sorghum, cereals, alfalfa, vegetables, potatoes, citrus fruits and olives	[38]
Evrotas	2,418	90	Oranges, olives, vine trees and vegetables	[39]
Louros	926	80	Citrus fruits, olives, corn, alfalfa and cotton	[37]
Loudias	1,638	38	Cotton, corn, rice, fruit trees (mainly peach and apples), sugar beet and vegetables	[34]

CA catchment area, *L* river length

Fig. 4 The Greek rivers that have been monitored for pesticide residues

The majority of the research activities have been carried out in Northern and Central Greece (Fig. 4) by research institutes (i.e. University of Ioannina, Aristotle University of Thessaloniki, etc.), national environmental institutes (i.e. General Chemical State Laboratory of Greece) and regional water catchment agencies. Research reports from the latter are often not readably accessible and, when available, they tend to collect general information; therefore, they have not been considered in this review.

The major Greek rivers monitored for pesticide residues are Aliakmon [40], Loudias [41], Axios [33], Pinios [42], Kalamas [43], Mornos [44], Evrotas [39], Acheloos [29], Evros [30], Strymonas [31], Ardas [30] and Erythropotamos [30] (Fig. 4). Among them, Aliakmon, Axios, Loudias, Louros, Arachthos, Evros, Acheloos and Kalamas were monitored more systematically.

The first systematic research work has been carried out by Albanis et al. that analysed organophosphorus, carbamate, organochlorine and triazine pesticide residues in the Kalamas River basin (Epirus region), for the period between September 1984 and November 1985 [43]. The detected compounds (azinphos methyl,

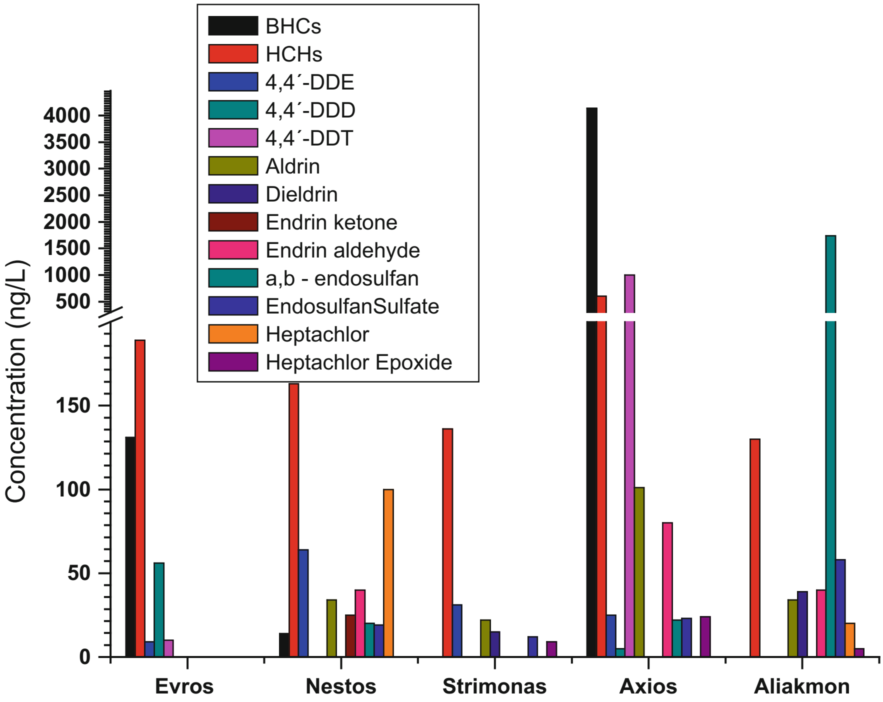


Fig. 5 Maximum concentration levels of organochlorine pesticides in various Greek rivers during the sampling period 1985–2005

parathion methyl, diazinon, carbofuran and carbaryl, lindane, atrazine, simazine and aminotriazole) were found to follow a seasonal pattern, with an increment during summer followed by a decrease during winter and an increase again during late spring. A lot of research has been conducted by the Albanis' group in the following years by performing systematic pesticide investigations in different river basins of Epirus and Macedonia regions. Between 1990 and 2006, Albanis and co-workers carried out pesticide monitoring studies in the following rivers: Kalamas [38], Arachthos [26, 37] and Louros in Epirus, Northwestern Greece [26, 45, 46]; Axios [47, 48], Loudias [34, 47, 48] and Aliakmonas in Macedonia, Northern Greece [34, 40]; and Evrotas in Peloponnese, Southern Greece [39]. Pesticide concentrations in the Axios River was also investigated by Mourkidou and co-workers [33], Kamarianos et al. [49], Golfinopulos et al. [36] and Miliadis and Malatou [42]. For other rivers of Central and Northeastern Greece, such as Aliakmonas, Loudias, Nestos, Strymonas and Evros, systematic investigations of pesticide occurrence were performed by different authors. For example, in Northeastern Greece, surface waters from three rivers, namely, Ardas, Evros and Erythropotamos, were studied by Vryzas et al. [30, 50] covering the distance from the Greek/Bulgarian borders down to the river's discharge (river's delta) in the Greek territory. Litskas et al. [31] conducted a monitoring study in the surface

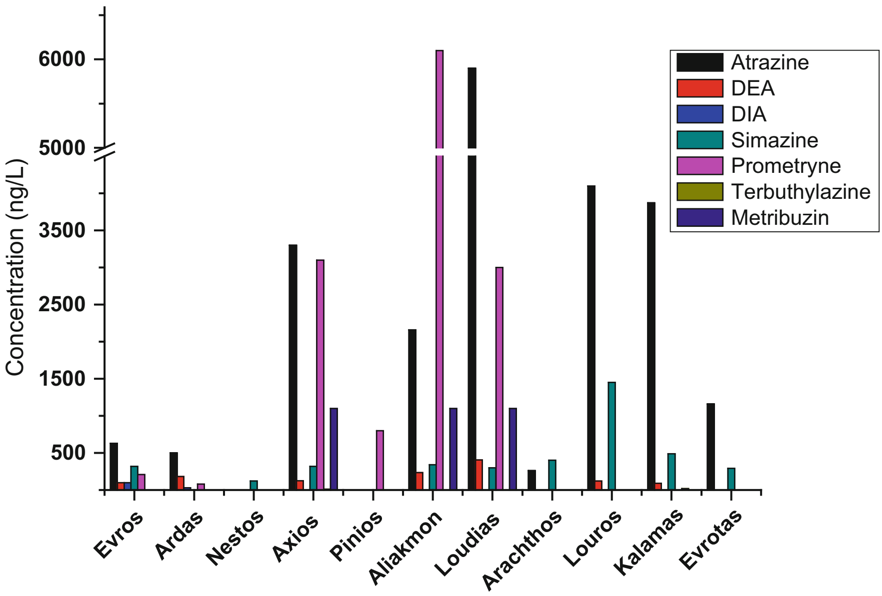


Fig. 6 Maximum concentration levels of triazine herbicides in various Greek rivers during the sampling period 1985–2005

waters of the Strymonas River catchment for several organic priority pollutants including organochlorine pesticides and their metabolites. In Western Greece, Stamatis et al. [29] conducted a 3-year monitoring campaign to evaluate pesticide contamination in surface waters of the Acheloos River, one of the most important water resources in Greece. For rivers located in Central Greece (Pinios and Asopos) and Peloponnese (Alphios), pesticide contamination was first recorded by Lekkas et al. [35, 51]; however, the levels detected are not discussed herein as they did not concern individual pesticide concentrations but ranges for all rivers monitored. Finally, for other Greek river catchment areas (i.e. Aposelemis, Mornos, Harvas, etc.), studies were sporadically conducted mainly during the 1990s [44, 52].

Compared to other European countries (Spain, England, Germany) [53], the studies of the occurrence of pesticides in Greek aquatic environment are rather limited. This is particularly true for studies focused on modern pesticides.

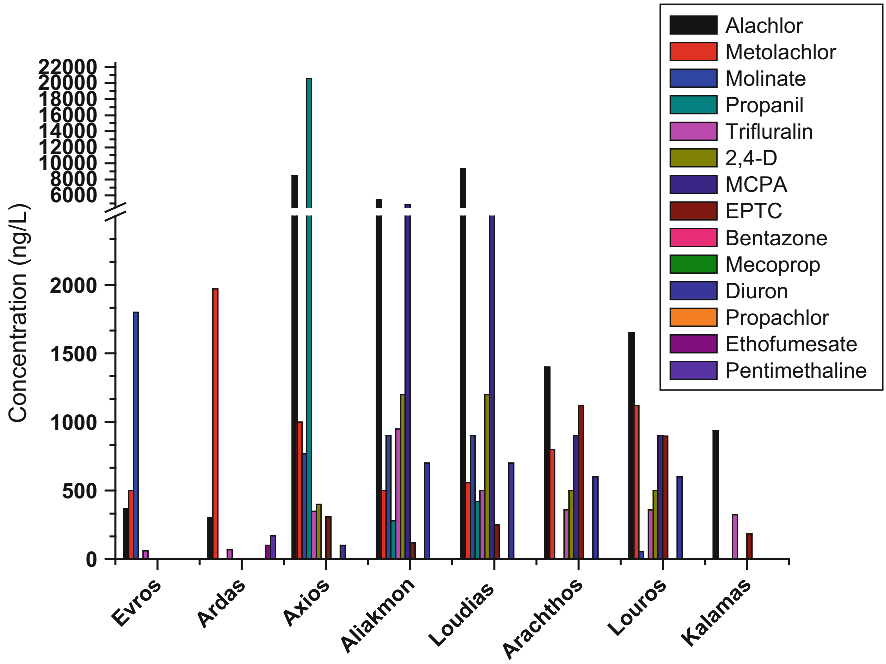


Fig. 7 Maximum concentration levels of other herbicides in various Greek rivers during the sampling period 1985–2005

7 Concentration Levels and Environmental Risk Assessment of Pesticide Residues in Greek Rivers

7.1 "Past" Investigations Between 1985 and 2005

7.1.1 Occurrence and Environmental Levels

In general, the pesticides that are most frequently encountered in river waters of Greece (Figs. 5, 6, 7 and 8) are those that were frequently used in agriculture, have lower K_{oc} values and show high or moderate environmental persistence.

The obtained results show that organochlorine pesticides (OPs) were among the most studied during the period of 1985–2005. Among them, dichlorodiphenyltrichloroethane (DDT) isomers and metabolites, hexachlorocyclohexane isomers (HCH), endosulfan, aldrin, chlordane, dieldrin, endrin, heptachlor, mirex, hexachlorobenzene (HCB), toxaphene and methoxychlor are classified as persistent organic pollutants (POPs). Organochlorine pesticide use has been banned in Europe since the mid-1970s, with the exception of lindane and endosulfan which were being used until June 2002 (Directive 2000/801/EC) [54] and 2007 (Directive 2005/864/EC) [55], respectively.

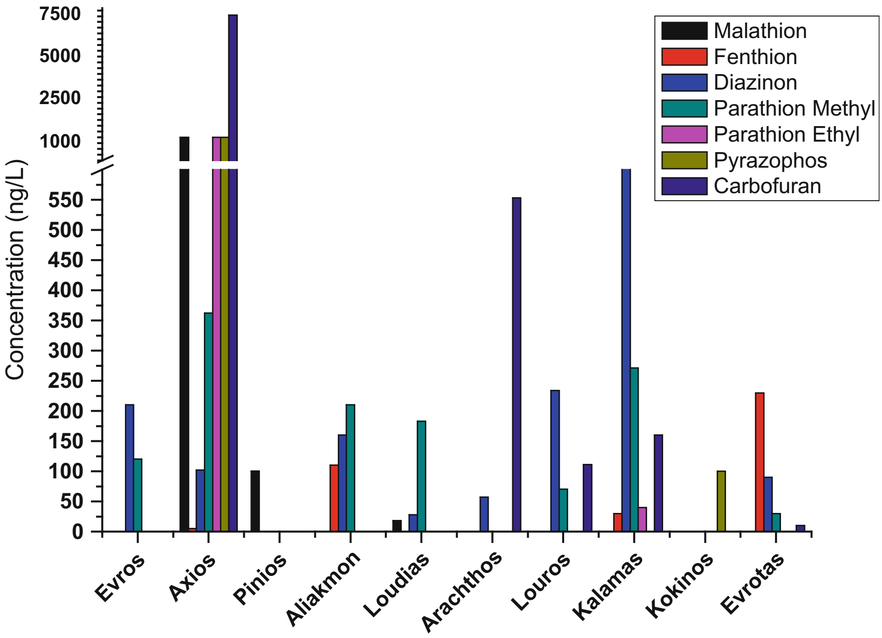


Fig. 8 Maximum concentration levels of organophosphorus insecticides in various Greek rivers during the sampling period 1985–2005

OP concentrations were generally very low, with the highest levels reported for hexachlorocyclohexane isomers (γ-HCH, 4.13 ng/L in Axios River) and a, b-endosulfan (1.74 ng/L in Aliakmon River), especially in rivers from Northern Greece (Fig. 5). For the transboundary rivers, Axios and Evros, the highest levels of all detected OPs were observed at the border sites, indicating probable transboundary pollution from the neighbouring countries [32, 36, 56]. In particular, the ubiquitous presence of lindane (γ-HCH) (occurrence 100%) in the Axios River, at sites located in the entrance of the river in the Greek territory, reveals that transboundary pollution is one of the major contributors in lindane contamination. This conclusion was also supported by the fact that lindane manufacturing was still active during the monitoring period in Skopje [32]. In addition, wet and dry deposition due to long-range transport from neighbouring regions or countries also contributed to the dispersion of lindane in surface waters across Greece [57–59].

Concerning herbicides, the triazine compounds, atrazine and simazine, and the chloroacetanilides, metolachlor and alachlor, have been most frequently detected in surface waters (Figs. 5 and 6). This was in accordance with their widespread and frequent use as well as their physicochemical properties that point to increased persistence and relatively high water solubilities [32, 46–48]. Similarly to OPs, higher triazine concentrations were detected in rivers from Northern Greece (e.g. Loudias River, alachlor 9.30 ng/L, atrazine 5.90 ng/L; Axios River, alachlor 5.50 ng/L; Aliakmon River, prometryne, 6.10 ng/L, alachlor 5.50 ng/L).

Other detected compounds in river waters were found in the following descending order: trifluralin, molinate, prometryne and propanil. Molinate and propanil were detected mainly in the Axios, Loudias and Aliakmon rivers, whose basins have very intensive rice cultivation activity (Fig. 7).

Particularly in the Axios River basin, several other pesticides (i.e. ethofumesate, bromopropylate, desmetryne, mevinphos, furalaxyl, cyanofos, cycloate, carbophenothion ethyl, terbumeton, atraton, coumaphos, napropamide, fluometuron, carbosulfan, methidathion, pirimiphos methyl and *cis*-permethrin) were detected in the time periods of 1993–1994 and 1997–1998, in which their patterns were highly consistent with their intensive use in cultivations such as cotton, corn, rice, fruit, tobacco and horticulture. However, they were encountered with trace concentrations in the majority of samples and at low detection percentages ($\leq 2\%$) [38].

The occurrence of insecticides was also investigated in Greek rivers (Fig. 8). Diazinon, methyl parathion and parathion were the most frequently encountered, followed by fenthion, carbofuran and malathion. The highest concentrations were recorded for malathion, parathion and pyrazophos (up to 2000 ng/L), parathion methyl (362 ng/L), carbofuran (7300 ng/L), diazinon (up to 775 ng/L) and fenthion (230 ng/L). The majority of the above peak values were encountered in the Axios River. In the Evrotas River, organophosphates were mainly detected, although in lower concentrations than herbicides. These findings were in agreement with the cultivation pattern of the drainage area of the Evrotas River, in which the main crops were oranges and olives with ploughed cultivations of vine trees and vegetables [39]. It is worthwhile to note that the application of parathion and parathion methyl in Greece has been prohibited since 2003 according to Directives 2001/520/EC and 2003/166/EC [60], and thus, residues of these compounds were not recorded in the most recent studies.

In the case of fungicides, the results from the reported studies show sporadic run-off of certain fungicides (captan, folpet) in the adjacent river water bodies. Only captan was monitored in the Loudias (maximum concentration, 24 ng/L) and Axios rivers (maximum concentration, 40 ng/L) and folpet (maximum concentration, 50 ng/L) in the Loudias River in low concentrations. These episodic occurrences of fungicides were related to their seasonal application in Greece. Results from the literature indicate that fungicide residues in the period 1985–2005 did not generally threaten contamination of freshwater and estuarine environments probably due to their low persistence [61].

As revealed from the corresponding studies, pesticide concentrations in river waters were higher during the period of their most intense application in spring and summer (May–August). Their presence at these time periods was also associated with their high surface run-off and the lower flow rates of the rivers [34, 37, 46, 47, 56].

The concentration levels depended also on several parameters such as degradation, dilution and meteorological conditions, mainly rainfall. For example, low concentrations were observed during the winter period due to increased rainfall events and extensive breakdown of pesticides after a long period from their application in mid-spring to early summer. Furthermore, in some cases, a lower peak is detected in late September–October because of the first rainfall events after the dry summer period [34, 38] that causes run-off inputs into the main river bodies.

7.1.2 Ecological Risk Assessment

Increased awareness for the environment has resulted in greater scrutiny for both "old" and "modern" pesticides and increased efforts have been directed to assess the potential effects and the ecotoxicological risk of these agrochemicals.

Environmental risk assessment involves the determination of pesticide exposure and adverse effects on nontarget organisms. The exposure assessment involves the measured environmental concentrations (MECs) derived either from pesticide monitoring studies or the predicted environmental concentration (PEC) that can be estimated by appropriate models incorporating several factors such as the rate of application, the environmental distribution and the bioaccumulation and persistence. Refinement of the exposure assessment, using measured rather than modelled exposure concentrations, has led to a more realistic risk assessment. The effect assessment involves the summary of toxicity reference values (TRVs) for the effects of target pesticides on selective representative organisms expressed as lethal concentration or effect concentration for the 50% of the organism population (LC_{50} or EC_{50}) or no observed effect concentration (NOEC).

The simplest method for the risk assessment is the calculation of the risk quotient (RQ), i.e. the quotient of the measured or estimated environmental concentration divided by a TRV value depending on the effect level (acute or chronic) [62]. The RQ method is useful only to rebut the presumption of potential adverse effects and belongs in the first stages or tiers of a risk assessment. For higher-tier assessments, the application of probabilistic approaches has been suggested. In probabilistic approaches, the risk is expressed as the degree of overlap between the exposure and the effects that is acceptable for a certain level of protection that would be attained. The level of certainty required in a particular situation is also taken under consideration [63]. Strengths of probabilistic approaches include the ability to quantify the magnitude and frequency of toxic effects and communicate more "meaningful" outputs to decision makers and the public. Potential weaknesses include the greater complexity, the lack of available expertise and guidance, difficulties in communicating results and the lack of established criteria for decision makers [64]. Other approaches include the scoring and ranking of pesticides into descriptive categories of risk in terms of their physicochemical and ecotoxicological properties. These methods are simple and fast for ecological screening assessments but are highly arbitrary [62].

In general, almost all Greek studies focused on risk assessment were performed by using the RQ method and only one has been conducted using the probabilistic risk assessment approach for the evaluation of negative impacts on ecosystems [65]. Therefore, in the next sections, the RQ methods used are described in detail and the outcomes are further discussed.

Risk Quotient Method (Deterministic: Tier 1)

For ecotoxicological risk assessment, the well-known risk quotient (RQ) or toxic unit method (*deterministic – tier 1*) for three taxonomic groups (i.e. algae, zooplankton, fish) at two effect levels (i.e. the acute level, using LC50 or EC50 values, and the chronic level, using PNEC values) was performed according to Directive 414/91/EEC [10].

For calculating the risk quotient (RQ) values, the reported concentrations of the pesticides in the surface waters of Greece were divided by an effect level (LC_{50} or EC_{50} or PNEC) reported in the literature according to the following equation:

$$\text{Risk } q\text{uotient } (\text{RQ}) = \frac{\text{exposure}}{\text{toxicity}} = \frac{\text{water } \quad c\text{oncentration}}{\text{LC}_{50}\,\text{or}\,\text{EC}_{50} \ \text{ or } \ \text{PNEC}}. \tag{1}$$

The initial approach to the risk assessment should be undertaken using the "worst-case" scenario. Since tier 1 screening is intended to be protective, the risk quotient was based on peak environmental concentrations, i.e. the maximum reported concentrations are used for the calculations [66]. This approach provides an estimate of the contribution of the compound of interest to the total toxicity of the water sample analysed to a certain taxonomic group (usually algae, *Daphnia magna* and fish). PNEC values were calculated by dividing the lowest long-term NOEC or short-term L(E)C50 (lethal/effect) when NOEC values are lacking, for the most sensitive species by the appropriate assessment factors (AFs) (Table 3) for three trophic levels (fish, zooplankton and phytoplankton) according to the European technical guidance document. Ecotoxicological data were obtained from the FOOTPRINT Pesticide Properties Database [68], the PAN Pesticides Database [69] and other studies containing toxicological data [70–73]. Risk quotient values were classified into descriptive categories of risk (levels of concern, LOCs) according to the classification presented in Hernando et al. [72]. For $RQ < 0.01$, $RQ = 0.01$, $RQ = 0.1$, $RQ = 1$ and $RQ > 1$, the categories negligible, low, medium, high and very high risk were labelled, respectively.

Table 3 Assessment factors to derive a $PNEC_{aquatic}$ [67]

Available data	Assessment factor
At least one short-term L(E)C50 from each of three trophic levels of the base set (fish, *Daphnia* and algae)	1,000[a]
One long-term NOEC (either fish or *Daphnia*)	100
Two long-term NOECs from species representing two trophic levels (fish and/or *Daphnia* and/or algae)	50
Long-term NOECs from at least three species (normally fish, *Daphnia* and algae) representing three trophic levels	10
Species sensitivity distribution (SSD) method	5–1
Field data or model ecosystems	Reviewed on a case-by-case basis

[a]A factor of 100 could be used for pesticides subject to intermittent release

Results and Discussion

Ecological Risk Assessment

The potential risk, according to RQ determination, for the reported maximum concentrations of pesticides in Greek freshwater systems to algae, zooplankton and fish community at acute effect level, is presented in Tables 4, 5 and 6. The pesticides that have been detected in surface waters, which showed negligible risk (RQ < 0.01), were not included in the tables. Results indicate that herbicide residues exhibit negligible acute toxicity to fish and invertebrates. However, they show medium to high toxicity to algae. This is due to the specific mode of action of herbicides that block the photosynthesis process. Similar results were found elsewhere [74] for a variety of aquatic plants including submerged macrophytes and algae, and they have been suggested as potential causes for losses of aquatic plants in streams or bays [75]. Primary production of aquatic plants and algae is the primary energy basis for aquatic ecosystems. Thus, herbicide impacts on primary producers are expected to have both direct and indirect impacts on the health of aquatic ecosystems. Generally, herbicides are more likely to reach high-risk scores for the aquatic environment due to their lower hydrophobicity and high algal toxicity [62].

From the detected herbicides in Greek rivers, atrazine, simazine, prometryne, alachlor, diuron, trifluralin, metribuzin, metolachlor and 2,4-D have shown a potential risk to algae according to the RQ method at acute effect level that ranged from low to medium to high (Table 4). The highest levels of risk were associated with residues of atrazine, prometryne and alachlor in rivers especially in the Aliakmon and Loudias rivers.

Not surprisingly, organophosphorus insecticides showed a higher risk compared to herbicides for *D. magna* (Table 5). Parathion methyl, parathion, diazinon, malathion, fenthion, ethion, carbofuran and azinphos methyl were the insecticides susceptible to risk. Risks ranged from low (malathion for the Loudias River) to very high (malathion and parathion for the Axios River). Only residues of aldrin, malathion and carbofuran show potential risk for the fish community in rivers of Northern Greece (Axios, Nestos, Evros, Strymonas) but at low to medium levels. As a general rule, for most organophosphorus insecticides, the risk seems to be higher in the short term due to high toxicity and relatively low persistence.

No risk was found to be associated with fungicides in Greek rivers due to the low concentration levels of these compounds that are probably related to the fact that fungicides are insufficiently persistent [61] and have relatively low toxicity.

A quotient addition approach assumes that toxicities are additive or approximately additive and that there are no synergistic, antagonistic or other interactions. The sum of the toxic quotients of all compounds detected gives an estimate of the total toxicity of the sample with respect to the compounds determined. This assumption may be most applicable when the modes of action of chemicals in a mixture are similar, but there is evidence that even with chemicals having dissimilar modes of action, additive or near-additive interactions are common [76]. It was not

Table 4 Potential risk of the reported concentration levels of pesticides (sampling period between 1985 and 2005) to the algae community of Greek freshwater systems according to risk quotient (RQ) determination [72]

Freshwater system	Levels of risk (acute effect level)					
	Low (RQ = 0.01)	Low to medium (0.01 < RQ<0.1)	Medium (RQ = 0.1)	Medium to high (0.1 < RQ < 1)	High (RQ = 1)	Very high (RQ > 1)
Rivers						
Aliakmon		Atrazine Metribuzin Diuron 2,4-D	Alachlor Trifluralin	Prometryne		
Loudias		Metribuzin Trifluralin Diuron 2,4-D		Atrazine Prometryne Alachlor		
Axios	Metolachlor	Atrazine Simazine Metribuzin Trifluralin Diuron Propanil 2,4-D				
Evros	Atrazine					
Kalamas	Alachlor	Trifluralin	Atrazine			
Arachthos	Metolachlor	Alachlor Trifluralin Diuron 2,4-D				

(continued)

Table 4 (continued)

Freshwater system	Levels of risk (acute effect level)					
	Low ($RQ = 0.01$)	Low to medium ($0.01 < RQ < 0.1$)	Medium ($RQ = 0.1$)	Medium to high ($0.1 < RQ < 1$)	High ($RQ = 1$)	Very high ($RQ > 1$)
Louros	Metolachlor	Alachlor Trifluralin Diuron Simazine 2,4-D	Atrazine			
Evrotas		Atrazine				
Pinios		Prometryne				

Table 5 Potential risk of the reported concentration levels of pesticides (sampling period between 1985 and 2005) to the zooplankton community (indicator organism: *Daphnia magna*) of Greek river systems according to risk quotient (RQ) determination [72]

Freshwater system	Levels of risk (acute effect level)					
	Low ($RQ = 0.01$)	Low to medium ($0.01 < RQ < 0.1$)	Medium ($RQ = 0.1$)	Medium to high ($0.1 < RQ < 1$)	High ($RQ = 1$)	Very high ($RQ > 1$)
Rivers						
Aliakmonas		Parathion methyl Fenthion	Diazinon			
Loudias	Malathion	Parathion methyl Diazinon				
Axios		Diazinon Parathion Methyl		Carbofuran		Malathion Parathion
Evros		Parathion methyl		Diazinon		
Kalamas		Parathion methyl		Diazinon		
Arachthos		Diazinon				
Louros		Parathion methyl		Diazinon		
Evrotas		Diazinon Fenthion				
Pinios		Parathion Malathion				

Table 6 Potential risk of the reported concentration levels of pesticides (sampling period between 1985 and 2005) to the fish community of Greek freshwater systems (indicator organism: rainbow trout) according to risk quotient (RQ) determination [72]

Freshwater system	Levels of risk (acute level)					
	Low ($RQ = 0.01$)	Low to medium ($0.01 < RQ < 0.1$)	Medium ($RQ = 0.1$)	Medium to high ($0.1 < RQ < 1$)	High ($RQ = 1$)	Very high ($RQ > 1$)
Rivers						
Axios	Carbofuran	Aldrin Malathion				
Evros		Aldrin				
Nestos	Aldrin					
Strymonas		Aldrin				
Aposelemis	Aldrin					

applicable to calculate the sum of toxic units from the data reported in the literature because in several studies the results are grouped according to seasons and there are no reported concentrations in details.

For dynamic systems such as rivers, the likelihood of long-term effects arising from the intermittent release of pesticides is lower than lakes and the principal risk being that of short-term toxic effects. Despite this general trend, chronic effects of pesticide residues on aquatic organisms were also evaluated [72] in order assess the negative impact on river ecosystems. According to the results, about twenty compounds showed high or very high risk for at least one of the three trophic levels. The results are briefly summarised as follows. The potential risk at chronic effect level for algae was found very high for *s*-triazines in rivers with the exceptions of the Honos, Havgas and Aposelemis rivers (Crete) and the lakes Mornos and Marathonas, where medium to high values were also observed. Additionally, several other herbicides such as alachlor, metolachlor, trifluralin, diuron and 2,4-D showed very high risk in the rivers that they had been detected. Compared to acute effect level, some organophosphorus (fenthion, parathion methyl) and organochlorine (lindane, aldrin, dieldrin) insecticides have shown that potential chronic risk ranged from low to medium to high levels. Very high risk was monitored for heptachlor in the Cretan and Evros rivers. On the contrary, organophosphorus insecticides show very high chronic risk for *D. magna* in all rivers except for fenthion in the Axios River, which showed medium to high risk. Generally, *s*-triazines and other herbicides showed low to medium risk for *Daphnia* with the exception of alachlor, trifluralin and MCPA that showed medium to high risk for the rivers of Northern Greece (Axios, Loudias, Aliakmonas). Very high risk was found only for 2,4-D in all rivers where it was detected, propanil in Axios and trifluralin in Aliakmonas. Organochlorines also presented chronic risk for *Daphnia* that ranged between low and medium to high level. Several compounds have shown very high risk for fish at the chronic effect level, i.e. atrazine, trifluralin, alachlor and propanil from the herbicides and parathion, malathion and carbofuran from the insecticides. For the rest of the compounds detected, the risk ranged between low and medium to high. It is worth noting that organochlorine insecticides such as aldrin, dieldrin, heptachlor and 4,4-DDT presented very high chronic effects for the fish in most of the rivers detected while medium to high risk was found for the rest of the cases. Finally, fungicides show negligible risk for algae, low to medium risk for *Daphnia* and medium to very high risk for fish.

Environmental Levels and Compliance with Environmental Quality Standard Requirements

Pesticide levels found in the Greek river basins were examined for their level of compliance with the EQS (Directive 2008/105/EC) [19]. By comparing herbicide and insecticide reported concentrations with maximum allowable concentration proposed by EQS (Table 1), we observed that many of the detected compounds exceed the limit values for most of the water bodies. Organophosphorus insecticide concentrations, especially, were significantly above the EQS; thus, they constitute a major threat to aquatic environments. In addition, non-compliance with the EQS was observed in some cases for the organochlorines aldrin (rivers Axios, Strymonas, Nestos, Evros and Aposelemis) and dieldrin (rivers Axios, Nestos and Aposelemis), for Σ_{DDTs} (rivers Axios, Strymonas, Nestos, Evros and Aposelemis), for endrin isomers (rivers Evros, Axios and Strymonas) and for lindane (rivers Aliakmon, Axios, Evros and Aposelemis).

7.2 "Recent" Investigations Between 2006 and 2015

7.2.1 Environmental Levels, Risk Assessment and Compliance with EQS

An important contribution in assessing modern trends of pesticide levels in Greek surface water has been recently provided by Vryzas et al. [30, 50], Karaouzas et al. [77] and the European Commission (EC) [20, 28]. In these recent studies, various groups of currently used pesticides were systematically investigated in different Greek river basins (Fig. 9) by incorporating in their results environmental risk assessment analysis, and they are presented in detail in the following paragraphs.

The detailed survey of Vryzas et al. [50] includes the monitoring of pesticides belonging to different classes in three sampling points along the river Erythropotamos, covering the distance from the Greek/Bulgarian borders down to the river's discharge (river's delta) in the Greek territory. In total, 13 sampling events were carried out from 2006 to 2007.

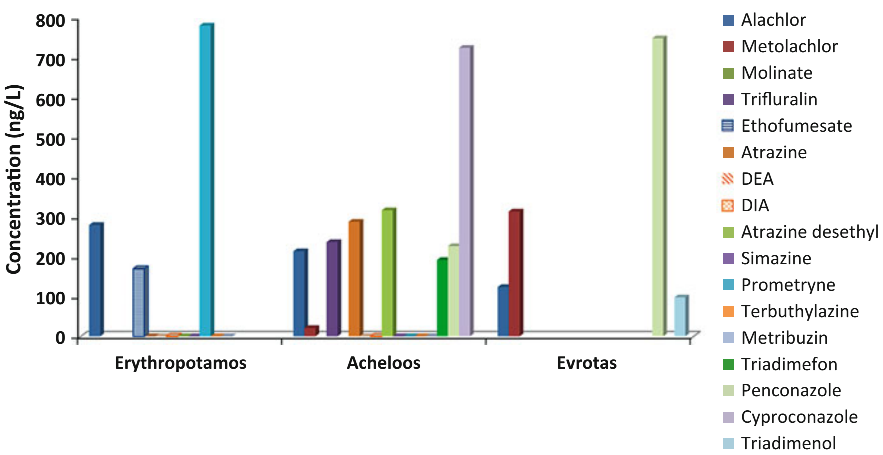


Fig. 9 Maximum concentration levels of herbicides and fungicides in various Greek rivers during the sampling period 2006–2015

Alachlor, metolachlor, prometryne and trifluralin were constantly detected in the Erythropotamos River. However, the concentrations of atrazine and its metabolites (DEA and DIA) were below the limit of detection, since the monitoring of Erythropotamos was conducted after atrazine's withdrawal from the Greek pesticide market (during 2006–2007). The authors emphasised the detection of *o,p*′-DDE and *o,p*′-DDT in the first sampling point near the Greek/Bulgarian borders. They concluded that the abundance of DDE over its parent compound in the surface water suggests contaminations from old usage rather than recent DDT input to the river and their presence could be attributed to the resuspension from sediments to water.

From the 17 compounds (pesticides and metabolites) that were detected in surface waters of the river, the soil-applied pesticides were the most frequently detected. High pesticide concentrations were detected within 2 months of their application, while extreme pesticide concentrations were detected in the beginning of the irrigation season or just after high-rainfall events. The most commonly encountered compounds in the river waters were atrazine, DEA, alachlor, trifluralin, prometryne, molinate, carbofuran, carbaryl and diazinon. Among them, concentrations of atrazine, metolachlor, alachlor, molinate and prometryne were frequently higher than 0.1 μg/L. Bifenthrin, carbofuran, diazinon, ethofumesate and *o,p*′-DDT rarely exceeded the level of 0.1 μg/L.

Aquatic risk assessment revealed nonacceptable risk for many of the detected compounds (especially for the group of insecticides) when extreme concentrations were used as PEC values.

In particular, for insecticides such as bifenthrin, chlorpyrifos ethyl, diazinon, parathion methyl and *o,p*′-DDE, very high RQ values were observed mainly due to their PNEC values on zooplankton.

The aforementioned results have been further confirmed by a second systematic study which was performed in riparian drainage canals of the transboundary river Erythropotamos, by the same research group during the period 2006–2008 [50]. The previously detected compounds such as alachlor, carbaryl, carbofuran, cypermethrin, diazinon, dimethoate, endosulfan, metolachlor, monilate, bifenthrin, prometryne and trifluralin were also encountered at a regular basis, whereas atrazine, DEA and DIA concentrations were below the limit of detection.

Aquatic risk assessment revealed nonacceptable risk for many compounds when median or extreme concentrations were taken into account. Similarly to the first study [30], the highest RQs were calculated for insecticides. For example, cypermethrin, endosulfan, methoxychlor, prometryne, pyrazophos, bifenthrin, chlorpyrifos, diazinon and λ-cyhalothrin showed very high RQ values mainly because of their relatively high toxicity to fish, algae or aquatic invertebrates. Lower RQs were calculated for herbicides, while the lowest was observed for fungicides.

Annual average concentrations of alachlor and atrazine (0.153, 0.505 μg/L were below the annual average EQS concentrations, while for chlorpyrifos, endosulfan and trifluralin, the corresponded values (0.118, 0.144 and 0.060 μg/L) surpassed the proposed annual average EQS concentrations. Finally, maximum concentrations of atrazine, chlorpyrifos and endosulfan were higher than the maximum allowable concentration proposed by EQS.

In the recent study of Karaouzas et al. [77], nutrients, trace metals and priority pesticide compounds were investigated for the first time in 12 stream sites distributed throughout the western part of the Evrotas River basin (Southeastern Greece) from 2006 to 2008. The catchment area of all selected sites vary from 1 to less than 10 km^2, apart from sites 5, 7 and 11 which have a catchment area of 20–25 km^2. Stream sites 1, 2, 3, 6 and 11 flow through the mountainous and semi-mountainous forested areas of the basin and are minimally or not at all affected by anthropogenic activities. Sites 7 and 9 are located in the peri-urban area of the city of Sparti, where numerous industrial units are situated, such as orange juice processing plants and meat processing factories. The other sites receive olive mill wastewaters (sites 2, 4, 5 and 12) during the olive harvesting period or flow through semi-forested areas (sites 2 and 12), olive tree fields (site 4) and urban (domestic) and agricultural areas (site 5).

No pesticides were detected in sites 1, 2, 6, 11 and 12, while site 8 was dry during all sampling campaigns and thus no samples of water were collected. Seven pesticides were detected in the rest of the sites (alachlor, metolachlor, penconazole, triadimenol, fenthion, dimethoate and malathion) thereby reflecting their abundant current use in the area and the river watersheds.

The fungicide penconazole was detected in all contaminated water samples (sites 7, 9 and 10) with concentrations ranging from 0.748 to 0.071 μg/L, as a result of intense applications in the field throughout the year. Its application includes gardens, vineyards, olive groves and agricultural peri-urban areas while it is relatively persistent in water (half-life in water is more than 706 days), which may cause chronic toxicity in aquatic life. The highest concentration was observed for dimethoate in site 9, at 5.58 μg/L. Since it has a low half-life time (7 days), this high concentration may be attributed to a recent application of this pesticide or spill-off. Triadimenol, an azole fungicide and a metabolite of triadimefon, was also detected at site 9, at a concentration of up to 0.098 μg/L. The priority compound alachlor was also detected at a concentration level of up to 0.124 μg/L, which is much lower than the maximum acceptable concentration (MAC) for this compound in inland surface waters (0.7 μg/L). Metolachlor, another extensively used herbicide, was detected at 0.314 μg/L. As far as insecticides are concerned, fenthion and malathion were sporadically encountered with mean concentrations of 0.06 and 0.548 μg/L, respectively.

Aquatic risk assessment showed high RQ for all the detected insecticides, thus suggesting probable adverse effects on the stream biota. For instance, malathion, dimethoate, penconazole and fenthion presented very high toxicity risk ($RQ > 1$). Similarly to the works of [30, 50], although fungicides and herbicides were generally more frequently detected in water samples than insecticides, their RQs were found to be lower than insecticides. Triadimenol showed medium risk, whereas the risk of toxic effects to stream organisms by alachlor and metolachlor remained low.

According to the findings of the study, it is apparent that small streams can receive pollution loads from point and diffuse sources that may not be detected in larger-scale monitoring. In this sense, this work reveals that the inclusion of streams with small catchment areas into WFD monitoring and assessment programmes is essential, especially those of the Mediterranean region.

Finally, in the very recent study of Stamatis et al. [29], a three-year survey (March 2005–February 2008) was conducted to investigate, on a monthly basis, the presence of pesticides belonging to various categories and transformation products in the Acheloos River. Acheloos, located in the southwestern part of the country (Western Greece), is one of the most important rivers, the first in water contribution and the second in length, found in the Greek territory. Its yearly outflow is estimated to be $7.8 \times 109\ m^3$ and its drainage basin covers a total area of 6,329 km^2. The shape of its basin is oblong with a maximum axis of 147 km length and 63 km width. The ecological importance of the estuary is high as it is connected to coastal lagoons which are under the protection of the Ramsar Convention. Finally, the delta plain belongs to the Natura 2000 site network. Its water is used in agriculture as well as for the generation of electricity. The watershed is not industrialised and agriculture contributes about 45% of the average income for the region [29].

Among the thirty target compounds, nineteen pesticides and transformation products, four herbicides (alachlor, atrazine, S-metolachlor, trifluralin), one metabolite (desethyl atrazine, DEA), nine insecticides (chlorpyrifos, chlorpyrifos methyl, diazinon, dichlorvos, dimethoate, fenthion, methidathion, parathion methyl and pirimiphos methyl), one metabolite (malaoxon) and four fungicides (cyproconazole, penconazole, pyrazophos and triadimefon) were detected in the water samples during the three-year monitoring campaign (2005–2007). The highest frequency of detection was observed for diazinon (78.6%), alachlor (50%), penconazole (43.2%) and DEA (69.3%) for the categories of insecticides, herbicides, fungicides and transformation products, respectively.

The highest concentrations of pesticides in the Acheloos waters surrounded by agricultural areas were dependent on meteorological and hydrological conditions, while their annual distribution of mean concentrations was strongly affected by the elimination of tobacco cultivation in 2006 (the main cultivation of the area for many decades). Seasonal variation showed in general higher mean concentrations for spring and summer compared to autumn and winter. Fourteen of the nineteen compounds studied showed significant differences of three-year mean concentration values in spring and summer compared with the other two seasons. The presence of the transformation product of atrazine, DEA, was also investigated, and it was encountered with measurable concentrations in the majority of samples. The desethyl atrazine-to-atrazine ratio (DAR) was found quite high due to the past uses of atrazine and the prolonged degradation of atrazine in soil and surface water. DAR values were lower in 2005, increased in 2006 and reached the highest values in 2007.

Environmental risk assessment approach showed high risk for six insecticides of the total nineteen compounds (chlorpyrifos, malaoxon, fenthion, pirimiphos methyl and chlorpyrifos methyl) in 2005 and one in 2007 (chlorpyrifos methyl). In agreement with the aforementioned studies, fungicides presented the lowest RQ values using both median and extreme MEC values for both sampling years (2005 and

2007). In order to assess the synergistic toxicity of pesticides in mixtures, the cumulative risk quotients were determined by grouping the detected pesticides in three subcategories based on their mode of action, namely, organophosphorus insecticides, herbicides and azole fungicides. Cumulative risk quotients for fungicides were always lower than risk values presenting acceptable risk. For herbicides and insecticides, a decreasing trend in cumulative risks was observed from 2005 to 2007, which is highly consistent with the abolition of the tobacco crop in 2006. The high cumulative risks obtained for insecticide mixtures (up to 115.5 in 2005) suggest substantial synergistic effects for this class of pesticides.

Finally, the annual average (AA) and the maximum allowed concentration (MAC) of six pesticides (atrazine, simazine, alachlor, trifluralin, diazinon and chlorpyrifos) included in the list of the 33 priority substances were in general (except for chlorpyrifos for the year 2005 and diazinon for the year 2007) lower than the concentration levels of EQS. Consequently, pesticide monitoring results showed a good compliance with WFD for the majority of them.

8 Concluding Remarks and Future Trends

In this chapter, the environmental concentrations of pesticide residues obtained by recent monitoring studies (2006–2015) were compared with historical data (1985–2005) gathered during previous monitoring investigations in key Greek river basins to describe trends in water quality status and determine potential adverse effects in the environment. Considering the data provided in this study, a summary of general remarks is highlighted below:

- Several pesticide compounds have been detected in river waters across Greece, indicating that some major water resources are contaminated.
- Within Greece, the contamination of freshwaters by pesticides follows similar concentration levels and patterns as reported in most European countries including Italy, Spain, France, the United Kingdom and Portugal.
- Research on pesticides in Greek rivers began with "old" pesticide groups including organochlorine, organophosphate and triazine compounds. However, recent studies have extended research beyond these target groups to include "modern" pesticide compounds such as carbamates, azoles, etc.
- Water pollution due to organochlorine pesticides showed that, despite their ban in Greek running waters, however, analysis of past and recent data showed a descending trend of OP levels in river waters between 1985 and 2015.
- The data provided in the period from 1985 to 2005 showed that the Axios, Aliakmon and Loudias were the rivers most polluted by pesticide residues based both on the number of detected compounds and the maximum detected concentrations.
- More herbicidal compounds than insecticides were detected in all the target monitoring periods. The most commonly encountered compounds include

atrazine, simazine, alachlor and metolachlor from herbicides and diazinon and malathion from insecticides. Generally, acetamide and triazine herbicides were widely used to control grasses and weeds in a broad range of crops and were detected at variable levels in Greek rivers.
- The data provided from the recent studies revealed that despite the fact that the levels of some compounds decreased with time (e.g. organochlorines, atrazine, alachlor, etc.), mainly due to a ban or the implementation of good agricultural practices, some river basins subjected to significant agriculture pressure continue to show high levels of pesticides.
- The countrywide patterns of chemical use are constantly changing, as the popularity of existing pesticide rises and falls and as new compounds are introduced into farming. With sufficient continued monitoring, the overall trend towards reduced pesticide loading into aquatic systems, caused by the combinations of use rate reductions, use restrictions and alterations of agricultural management practices, may become discernible.
- According to the ecological risk assessment, the risk ranged from negligible to high depending on the pesticide and the target organism. Herbicides showed low to high risk for the algae while negligible risk for invertebrates (*D. magna*) and fish (rainbow trout), whereas insecticides show negligible risk for the algae and low to high risk for *Daphnia*. Atrazine, alachlor, prometryne, metribuzin, trifluralin and 2,4-D were the herbicides that presented the higher risk and parathion, parathion methyl, fenthion, malathion, diazinon and carbofuran were the insecticides with the higher risk.
- Considering the calculated RQ values, in some cases, the environmental concentrations of pesticides may be shown to be non-compliant with the EQS (Directive 2008/105/EC) and therefore present a risk according to these standards to aquatic environment.

Despite the main conclusions described above, there is still a considerable need for future research and other activities that must be addressed in this area. The most important research topics from our point of view are the following:

- Although the existing data cover the last three decades, only few of them concern annual monitoring surveys that include all pesticide categories. In addition, there is little consistency in the majority of the aforementioned studies in terms of site selection strategy, sampling methodologies, collection time and duration, selected analytes, analytical methods and detection limits. Thus, a harmonisation in the sampling strategy and the monitoring practices in accordance with the WFD is required. For future campaigns, specific insight into agricultural treatments and land use in the catchment could contribute to optimised water monitoring.
- Considering the fact that current information on environmental occurrence of pesticides is limited to certain rivers situated mainly in the north and west part of Greece, further work is needed to attain a more comprehensive picture of its occurrence and distribution in other significant rivers located in Central Greece (e.g. Sperheios, Pinios, etc.).

- Monitoring focused on certain key regions should be continued in order to better understand the effects of mitigation methods and changes in land uses.
- New analytical methods should be turned to multiclass methods in order to face up the large number of pesticides and their transformation products that should be screened.
- It is clear from the monitoring data that aquatic organisms were subjected to episodic pulses of varying frequency and magnitude. Thus, research is needed to characterise the response of aquatic organisms to pulsed exposures.
- Only very few transformation products, mainly those of “old” or “banned” pesticides, were examined in the previous nationwide investigations. Therefore, to understand the wider picture, transformation products of broad spectrum and demonstrably relevant pesticides found in surface waters should be included in future monitoring studies. In addition, incorporation of concentrations and toxicological data of the most important pesticide transformation products in the RQ method would have afforded a better assessment of the ecological risk and its temporal variation.
- Although the risk quotient method is attractive for its relative simplicity, higher-tier risk assessments should be performed and further validated in well-designed microcosm or field studies in order to draw conclusive results on the potential adverse effects on river ecosystems.

Acknowledgements This research has been financed by the Hellenic Centre for Marine Research – (HCMR), through the research programme “Analysis of aqueous samples for the determination of pesticide residues in river network of Messenia area”, research grant no. 87093, which is gratefully appreciated.

References

1. Mehrpour O, Karrari P, Zamani N, Tsatsakis AM, Abdollahi M (2014) Occupational exposure to pesticides and consequences on male semen and fertility: a review. Toxicol Lett 230 (2):146–156
2. Pal A, He Y, Jekel M, Reinhard M, Gin KY-H (2014) Emerging contaminants of public health significance as water quality indicator compounds in the urban water cycle. Environ Int 71:46–62
3. Luo Y et al (2014) A review on the occurrence of micropollutants in the aquatic environment and their fate and removal during wastewater treatment. Sci Total Environ 473–474:619–641
4. Nowell LH, Norman JE, Moran PW, Martin JD, Stone WW (2014) Pesticide Toxicity Index–a tool for assessing potential toxicity of pesticide mixtures to freshwater aquatic organisms. Sci Total Environ 476–477:144–157
5. Müller K, Magesan GN, Bolan NS (2007) A critical review of the influence of effluent irrigation on the fate of pesticides in soil. Agric Ecosyst Environ 120(2-4):93–116
6. Arias-Estévez M, López-Periago E, Martínez-Carballo E, Simal-Gándara J, Mejuto J-C, García-Río L (2008) The mobility and degradation of pesticides in soils and the pollution of groundwater resources Agriculture. Ecosyst Environ 123(4):247–260
7. Flury M (1996) Experimental evidence of transport of pesticides through field soils – a review. J Environ Qual 25(1):25–45

8. Lapworth DJ, Baran N, Stuart ME, Ward RS (2012) Emerging organic contaminants in groundwater: a review of sources, fate and occurrence. Environ Pollut 163:287–303
9. European Crop Protection Association (ECPA) Annual Review 2011–2012. http://www.ecpa.eu/files/attachments/Annual%20Report_web.pdf
10. European Commission (EC) (1991) Council Directive 91/689/EEC of 12 December 1991 on hazardous waste OJ L 377
11. European Commission (EC) (1998) Commission Directive 98/38/EC of 3 June 1998 adapting to technical progress Council Directive 74/151/EEC on certain components and characteristics of wheeled agricultural or forestry tractors OJ L 170/13
12. European Commission (EC) (2006) Directive 2006/12/EC of the European Parliament and of the Council of 5 April 2006 on waste OJ L 114/9
13. European Commission (EC) (1991)Council Directive 91/414/EEC of 15 July 1991 concerning the placing of plant protection products on the market OJ L 230/ 1
14. European Commission (EC) (2009) Directive 2009/128/EC of the European Parliament and of the Council of 21 October 2009 establishing a framework for Community action to achieve the sustainable use of pesticides. OJ L 309/71
15. European Commission (EC) (1998) Directive 98/8/EC of the European Parliament and of the Council concerning the placing of biocidal products on the market OJ L 123/1
16. European Commission (EC) (2006) Directive 2006/118/EC of the European Parliament and of the Council of 12 December 2006 on the protection of groundwater against pollution and deterioration OJ L 372/19
17. European Commission (EC) (2000) Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy OJ L 327/1
18. European Commission (EC) (2001) Decision No 2455/2001/EC of the European Parliament and of the Council of 20 November 2001 establishing the list of priority substances in the field of water policy and amending Directive 2000/60/EC OJ L 331/1
19. European Commission (EC) (2008) Directive 2008/105/EC of the European Parliament and of the Council of 16 December 2008 on environmental quality standards in the field of water policy, amending and subsequently repealing Council Directives 82/176/EEC, 83/513/EEC, 84/156/EEC, 84/491/EEC, 86/280/EEC and amending Directive 2000/60/EC of the European Parliament and of the Council. OJ L 348/84
20. European Commission (EC) (2013) Directive 2013/39/EU of the European Parliament and of the Council of 12 August 2013 amending Directives 2000/60/EC and 2008/105/EC as regards priority substances in the field of water policy OJ L 226/1
21. Stockholm Convention on Persistent Organic Pollutants (POPs) (2004) United Nations Environment Programme (UNEP) Chemicals. Geneva. http://chm.pops.int/Home/tabid/2121/mctl/ViewDetails/Event-ModID/871/EventID/230/xmid/6921/Default.aspx
22. Vasiljević T, Dujaković N, Radišić M, Grujić S, Laušević M, Dimkić M (2012) Methods for monitoring of pesticide residues in water: current status and recent trends. Water Sci Technol 66(5):965–975
23. Lambropoulou DA (2013) Microextraction techniques coupled to advanced GC–MS techniques for analysis of environmental samples. In: Imma F, Thurman EM (eds) Comprehensive analytical chemistry. vol 61. Elsevier, pp 23–54
24. Duca RC, Salquebre G, Hardy E, Appenzeller BM (2014) Comparison of solid phase- and liquid/liquid-extraction for the purification of hair extract prior to multi-class pesticides analysis. J Chromatogr B Anal Technol Biomed Life Sci 955–956:98–107
25. Lambropoulou DA, Konstantinou IK, Albanis TA (2007) Recent developments in headspace microextraction techniques for the analysis of environmental contaminants in different matrices. J Chromatogr A 1152(1–2):70–96
26. Albanis TA, Hela DG, Lambropoulou DA, Sakkas VA (2004) Gas chromatographic mass spectrometric methodology using solid phase microextraction for the multiresidue determination of pesticides in surface waters (N.W. Greece). Int J Environ Anal Chem 84:1079–1092

27. European Commission (EC) (2002) Commission Decision of 12 August 2002 implementing Council Directive 96/23/EC concerning the performance of analytical methods and the interpretation of results OJ L 221/8
28. SANCO/12571/2013 (2014) Guidance document on analytical quality control and validation procedures for pesticide residues analysis in food and feed
29. Stamatis N, Hela D, Triantafyllidis V, Konstantinou I (2013) Spatiotemporal variation and risk assessment of pesticides in water of the lower catchment basin of Acheloos River, Western Greece. Sci World J doi:10.1155/2013/231610
30. Vryzas Z, Vassiliou G, Alexoudis C, Papadopoulou-Mourkidou E (2009) Spatial and temporal distribution of pesticide residues in surface waters in northeastern Greece. Water Res 43 (1):1–10
31. Litskas VD, Dosis IG, Karamanlis XN, Kamarianos AP (2012) Occurrence of priority organic pollutants in Strymon river catchment, Greece: inland, transitional, and coastal waters. Environ Sci Pollut Res Int 19(8):3556–3567
32. Papadopoulou-Mourkidou E (2002) Quality of surface waters of Macedonia-Thrace, Northern Greece, Quality control program, Final Report. Ministry of Agriculture, Thessaloniki (in Greek)
33. Papadopoulou-Mourkidou E, Karpouzas DG, Patsias J et al (2004) The potential of pesticides to contaminate the groundwater resources of the Axios river basin in Macedonia, Northern Greece. Part I. Monitoring study in the north part of the basin. Sci Total Environ 321:127–146
34. Albanis TA, Hela DG, Sakellarides TM, Konstantinou IK (1998) Monitoring of pesticide residues and their metabolites in surface and underground waters of Imathia (N. Greece) by means of solid-phase extraction disks and gas chromatography. J Chromatogr A 823:59–71
35. Lekkas T et al (2004) Evaluation of the pollution of the surface waters of Greece from the priority compounds of list II, 76/464/EEC directive, and other toxic compounds. Environ Int 30(8):995–1007
36. Golfinopulos SK, Nikolaou AD, Kostopoulou MN, Xilourgidis NK, Vagi MC, Lekkas DT (2003) Organochlorine pesticides in the surface waters of Northern Greece. Chemosphere 50:507–516
37. Albanis TA, Danis TG, Hela DG (1995) Transportation of pesticides in estuaries of Louros and Arachthos rivers. Sci Total Environ 171:85–93
38. Lambropoulou DA, Sakkas VA, Hela DG, Albanis TA (2002) Application of solid phase microextraction (SPME) in monitoring of priority pesticides in Kalamas River (N.W. Greece). J Chromatogr A 963:107–116
39. Angelidis MO, Markantonatos PG, Bacalis NC, Albanis TA (1996) Seasonal fluctuations of nutrients and pesticides in the basin of Evrotas river, Greece. J Environ Sci Health A 31 (2):387–410
40. Albanis TA, Danis TG, Voutsa D, Kouimtzis T (1995) Evaluation of chemical parameters in Aliakmon river northern Greece. Part III Pesticides. J Environ Sci Health A 30(9):1945–1956
41. Albanis TA, Danis TG, Kourgia MK (1994) Transportation of pesticides in estuaries of the axios, Loudias and Aliakmon rivers (Thermaikos Gulf), Greece. Sci Total Environ 156(1):11–22
42. Miliadis GE, Malatou PT (1997) Monitoring of the pesticide levels in natural waters of Greece. Bull Environ Contam Toxicol 59:917–923
43. Albanis TA, Pomonis PJ, Sdoukos TA (1986) Organophosphorus and carbamates pesticide residues in the aquatic system of Ioannina basin and Kalamas river (Greece). Chemosphere 15:1023–1034
44. Tsipi D, Hiskia A (1996) Organochlorine pesticides and triazines in the drinking water of Athens. Bull Environ Contam Toxicol 31:387–410
45. Albanis TA, Hela DG (1995) Multiresidue analysis of pesticides in various environmental water simulated samples using solid phase extraction disks followed by gas chromatography techniques with FTD and MSD. J Chromatogr A 707:283–292

46. Albanis TA, Hela DG (1998) Pesticide concentration in Louros river and their fluxes into marine environment. Int J Environ Anal Chem 70:105–120
47. Albanis TA (1991) Run-off losses of herbicides EPTC, molinate, simazine, diuron, propanil and metolachlor in Thermaikos Gulf. N Greece. Chemosphere 22:645–653
48. Albanis TA (1992) Herbicide losses in run-off from the agricultural area of Thessaloniki in Thermaikos Gulf. Sci Total Environ 114:59–71
49. Kamarianos A, Karamanlis X, Galoupi E (2002) Pollution of coastal areas of N. Greece by organochlorine pesticides and polychlorinated biphenyls (PCBs). In: Nikolaou K (ed) Proceedings of the 1st environmental conference of Macedonia Thessaloniki, Greece, pp 116–121
50. Vryzas Z, Alexoudis C, Vassiliou G, Galanis K, Papadopoulou-Mourkidou E (2011) Determination and aquatic risk assessment of pesticide residues in riparian drainage canals in northeastern Greece. Ecotoxicol Environ Saf 74(2):174–181
51. Lekkas T et al (2003) Monitoring priority substances of directives 76/464/EEC and 2000/60/EC in Greek water bodies. J Environ Monit 5:593–597
52. Vassilakis I, Tsipi D, Scoullos M (1998) Determination of a variety of chemical classes of pesticides in surface and ground waters by off-line solid-phase extraction, gas chromatography with electron-capture and nitrogen-phosphorus detection, and high-performance liquid chromatography with post-column derivatization and fluorescence detection. J Chromatogr A 823:49–58
53. Loos R, Gawlik BM, Locoro G, Rimaviciute E, Contini S, Bidoglio G (2009) EU-wide survey of polar organic persistent pollutants in European river waters. Environ Pollut 157(2):561–568
54. Decision 2000/801/EC concerning the non-inclusion of lindane in Annex I to Council Directive 91/414/EEC and the withdrawal of authorizations for plant protection products containing this active substance. Off J Eur Communities 324:42–43
55. Decision 2005/864/EC concerning the non-inclusion of endosulfan in Annex I to the Council Directive 91/414/EEC and the withdrawal of authorizations for plant protection products containing this active substance. Off J Eur Communities 317:37–40
56. Angelides MO, Albanis TA (1996) Pesticide residues and heavy metals in the Evros river delta, N.E. Greece. Toxicol Environ Chem 53:33–34
57. Beyer A, Mackay D, Matthies M, Wania F, Webster E (2000) Assessing long-range transport potential of persistent organic pollutants. Environ Sci Technol 34:699–703
58. Cleemann M, Poulsen ME, Hilbert G. Deposition of lindane in Denmark. Chemosphere 30:2039–2049
59. Van Pul WAJ, De Leeuw FAAM, Van Jaarsveld JA, Van der Gaag MA, Sliggers CJ (1998) The potential for long-range transboundary atmospheric transport. Chemosphere 37:113–141
60. European Commission (EC) (2003) Technical guidance document in support of commission directive 93/67/EEC on risk assessment for new notified substances and commission regulation (EC)No 1488/94 on risk assessment for existing substances, part II
61. Readman JW, Albanis TA, Barcelo D, Galassi S, Tronczynski J, Gabrielides GP (1997) Fungicide Contamination of Mediterranean estuarine waters: results from a MED POL pilot survey. Mar Pollut Bull Environ Contam Toxicol 34:259–263
62. Finizio A, Calliera M, Vighi M (2001) Rating systems for pesticide risk classification on different ecosystems. Ecotoxicol Environ Saf 49:262–274
63. Solomon K, Giesy J, Jones P (2000) Probabilistic risk assessment of agrochemicals in the environment. Crop Prot 19:649–655
64. Hart A (2001) Probabilistic risk assessment for pesticides in Europe: implementation & research needs. A Report from the European workshop on probabilistic risk assessment for the environmental impacts of plant protection products. The Netherlands
65. Hela DG, Lambropoulou DA, Konstantinou IK, Albanis TA (2005) Determination of pesticide residues and ecological risk assessment in lake Pamvotis (N.W. Greece). Environ Toxicol Chem 24:1548–1556

66. Steen RJCA et al (1999) Ecological risk assessment of agrochemicals in European estuaries. Environ Toxicol Chem 18:1574–1581
67. European Commission (EC) (2002) Technical Guidance Document on risk assessment in support of Council Directive 93/67/EEC on risk Assessment for new notified substances and Commission Regulation (EC) 1488/94 on risk assessment for existing substances and Directive 98/8/EC of the European Parliament and the Council concerning the placing of biocidal products of the market
68. Pesticide Action Network, Pesticide Database (PAN) http://www.pesticideinfo.org/
69. Pesticide Properties database (PPDB). In: Agriculture & Environment Research Unit (AERU), University of Hertfordshire. http://sitem.herts.ac.uk/aeru/footprint/index2.htm
70. Konstantinou I, Hela D, Lambropoulou D, Albanis T (2008) Monitoring of pesticides in the environment. In: Tadeo (eds) Analysis of pesticides in food and environmental samples. CRC, pp 319–358
71. Thomatou A-A, Zacharias I, Hela D, Konstantinou I (2013) Determination and risk assessment of pesticide residues in lake Amvrakia (W. Greece) after agricultural land use changes in the lake's drainage basin. Int J Environ Anal Chem 93(7):780–799
72. Hernando MD, Mezcua M, Fernández-Alba AR, Barceló D (2006) Environmental risk assessment of pharmaceutical residues in wastewater effluents, surface waters and sediments. Talanta 69(2):334–342
73. Environmental Protection Agency (USEPA) (1998) Guidelines for ecological risk assessment. Fed Reg 63:26846–26924
74. Mayer FL, Ellersieck MR (1986) Manual of acute toxicity: interpretation and data base for 410 chemicals and 66 species of freshwater animals. US Department of the Interior, Fish and Wildlife Service, Washington, DC
75. Kemp WM, Boynton WR, Cunningham JJ, Stevenson JC, Jones TW, Means JC (1985) Effects of atrazine and linuron on photosynthesis and growth of the macrophytes. *Potamogeton perfoliatus* L. and *Myriophyllum spicatum* L. in an estuarine environment. Mar Environ Res 16(4):255–280
76. Broderius SJ, Kahl MD, Hoglund MD (1995) Use of joint toxic response to define the primary mode of toxic action for diverse industrial organic chemicals. Environ Toxicol Chem 9:1591
77. Karaouzas I, Lambropoulou DA, Skoulikidis NT, Albanis TA (2011) Levels, sources and spatiotemporal variation of nutrients and micropollutants in small streams of a Mediterranean River basin. J Environ Monit 13(11):3064–3074

Geochemical Processes of Trace Metals in Fresh–Saline Water Interfaces. The Cases of Louros and Acheloos Estuaries

Michael Scoullos and Fotini Botsou

Abstract Fresh–saline water interfaces are sites of major transformations on the speciation and the distribution of trace metals, through complex processes. The present chapter considers trace metal geochemical processes at fresh–saline water interfaces of representative Greek riverine systems, namely of those of the perennial medium-sized Louros River and the big and highly fragmented Acheloos River. Dissolved and particulate metals, as well as metal fractions in the sediments, are considered in combination with physicochemical parameters, and mineral magnetic measurements are used for tracing the origin of particle populations (lithogenic, anthropogenic, authigenic), and their compositional alterations during their passage from the rivers, through the interfaces, to sea. The interfaces of the two systems have distinct characteristics both on a spatial and a temporal scale, thus allowing for a diversity of trace metal behaviour patterns to emerge. In the small, perennial Louros system, trace metals are trapped within the thin, yet stable salt wedge. In the heavily fragmented Acheloos system, variations of the water and sediment discharges have moved the active interface landwards, where due to the reduction of dilution effects by inert, detrital particles, the fingerprint of the authigenic and anthropogenic component of trace metals has become more pronounced. The results of the research carried out in the two distinctive fresh–saline water interface systems are important not only in order to enlighten us about the geochemical processes in nature, but also in order to provide the necessary knowledge to properly manage these systems for the benefit of the environment and the sustainable development of the impacted areas.

Keywords Distribution coefficient, Estuarine mixing, Salt wedge, Sorption–desorption processes, Trace metals

M. Scoullos (✉) and F. Botsou
Laboratory of Environmental Chemistry, Department of Chemistry, National and Kapodistrian University of Athens, Panepistimioupolis Zografou, 157 84 Athens, Greece
e-mail: scoullos@chem.uoa.gr

N. Skoulikidis et al. (eds.), *The Rivers of Greece: Evolution, Current Status and Perspectives*, Hdb Env Chem (2018) 59: 241–278, DOI 10.1007/698_2016_469,

Contents

1 Introduction

Fresh–saline water interfaces, including a variety of systems such as river mouths, estuaries, ríás and coastal lagoons, are recipients of major discharges of trace metals deriving from land runoff, industrial and urban discharges and atmospheric precipitates (e.g. [1–3]). The aforementioned systems are characterised as "chemical reactors" wherein, under strong hydrodynamic and physicochemical gradients, complex heterogeneous processes greatly affect the distribution of trace metals, and eventually, the fluxes of metals that reach the adjacent sea [4–7].

The geochemical dynamics of fresh–saline water interfaces is influenced by the specific, physical and climatic conditions that control the discharge rates and the residence time of trace metals, the geomorphological conditions that affect the overall structure of these systems, and the numerous and complex biogeochemical processes that define the distribution of trace metals over the particulate and dissolved phase, thus the composition of the deposited sediment. The biogeochemical processes include complexation reactions with dissolved organic and inorganic ligands, adsorption/desorption reactions onto inorganic and organic suspended particles, flocculation and coagulation of colloidal and particulate species and remobilisation from sediments. All these processes vary, depending on pH, ionic strength, the amount and the composition of suspended particles, as well as with redox conditions [5, 8, 9].

The changes in the distribution between the dissolved and the particulate phase are demonstrated through the addition, or the removal of dissolved trace metals (e.g. [4, 6, 7]). Sediments in such transitional systems can act in some cases as sinks, and in some other cases as secondary sources of metals for the adjacent marine environment. The character of their specific function is defined by complex physical, geochemical and biological factors [10–12] and might change periodically.

In the riverine fresh–saline water interfaces perhaps the most significant physical factor is the energy of the overlying flow. Strongly dependent on the flow, both in terms of volume and velocities, are the residence time of waters and suspended particulate matter at the fresh–saline water interfaces [13]. In the stratified estuary of the big Rhone River (drainage basin: 98,800 km^2) with an average water discharge of 1,700 m^3/s that reaches more than 8,000 m^3/s at flood events [2], the rapid flushing of the water and a large fraction of particulate matter in the brackish surface plume (in a matter of a few days) reduces the quantity of suspended matter available for exchanges between the dissolved and particulate phase, as well as the contact time between the particles and the solution. In this case, dilution effects predominate over adsorption/desorption processes [14]. Of course, besides the kinetic control (rates of chemical reactions in relation to estuarine size and hence freshwater retention time) thermodynamic considerations are also important: re-suspension of sediments driven by strong tidal currents or even wind stress, and landward movement of suspended particles depleted in adsorbed metals, or in situ addition of particles of biogenic origin, could generate disequilibrium with respect to particle-water metal exchanges, thus enhancing solution–particle interactions, even in small estuarine systems [13, 15, 16].

Other factors interlinked to the prevailing hydrological regime are the dilution effects (e.g. [2, 17]), changing of the redox status of waters and/or sediments (e.g. [5, 18]) and benthic fluxes [4, 19], re-suspension of sediments [20], as well as the granulometry and the composition of suspended particulate matter (SPM) (e.g. [21]). The particle size distribution and the composition of SPM (i.e. organic, inorganic, inert minerals and highly reactive Fe/Mn oxyhydroxides, complex surfaces or flocculated aggregate constituents) are fundamental properties for the reactivity of trace metals during estuarine mixing, settling velocities and re-suspension potential within the mixing zone, and dispersion pathways beyond the mixing zone [3, 16, 17, 22].

In the Mediterranean basin, apart from a few large, perennial rivers, with catchment areas larger than 20,000 km^2, such as the Rhone, the Ebro (Spain), the Nile, the Evros (Greece) and Po, there are hundreds of medium ($<5,000$ km^2) and small (<500 km^2) streams, many of which are intermittent or ephemeral, representing ~12% of the Mediterranean's drainage basin. This figure rises up to ~42.5% if the Nile river basin is excluded [23]. In the medium/small rivers, the low river flow and the absence of strong tidal currents result in the suppression of the mixing zone to only a few kilometers, in contrast, for example, to the macrotidal large Scheldt Estuary that extents over 100 km and has a residence time of two to three months depending on the seasonal variations of the river flow [18, 19]. In these systems, river discharge, rather than marine influences, plays the dominant role on the stratification of the estuary and the advance or retreat of the salt wedge [24].

A very important feature of the river flow, particularly in the Mediterranean, is the high variability which exceeds the margins of seasonality, not only because of water scarcity linked to the semi-arid character of the region and other natural phenomena (e.g. the enhanced frequency of droughts and floods attributed to climate change), but also due to severe anthropogenic interventions, such as dams, water abstraction and flow diversions. These changes have a direct impact on water and sediment flows and result in a series of complex indirect implications on the behaviour of trace metals. For example, the reduction of the Strymon River inflows into the N. Aegean Sea, by approximately 30% due to extensive irrigation and reduced precipitation, favoured the development of a salt wedge intruding the upper part of the estuary [24].

Due to the unique combination of the physical, geomorphological and biogeochemical factors in each system, there is no common pattern of trace metals behaviour in fresh–saline water interfaces. Thus, despite the wealth of literature reports for the better understanding of the geochemical dynamics and processes in estuarine systems, further research on their structure and on the distributions of trace metals in their waters and sediments is still needed.

The present chapter elaborates on a number of important aspects of trace metal transport from land to the sea and the transformations occurring at the fresh–saline water interfaces that emerged from the examination of specific cases studied in the Laboratory of Environmental Chemistry (LEC) of the University of Athens, Greece, under the supervision of the first author who also introduced Chemical Oceanography to Greece in the late 1970s. Since then, multiple projects have been carried out by LEC scientists, in many active fresh/saline water interface systems ([25–31]), and valuable insights on the behaviour of trace metals in these systems have been gained.

The present work focuses on two riverine systems, namely those of rivers Louros and Acheloos, relying to a large extent on the work published by Scoullos et al. [28] and Dassenakis et al. [25, 32], respectively. These systems have very distinct characteristics with respect to their flows and sediment discharges, the interventions along their river courses, as well as the activities hosted in the respective catchment areas. The conditions of formation, as well as the surface of the fresh–saline water interface, vary widely among the systems primarily due to the diversity in their geomorphological features and hydrological regimes. In the present review, the selected systems demonstrate the diversity of patterns of trace metal behaviour and the conclusions drawn can be readily applicable to numerous comparable systems of the Mediterranean coastline and beyond.

2 Description of the Systems

The drainage basins of the Louros and Acheloos Rivers occupy an area of 785 km^2 [33] and 6,478 km^2 [34], respectively. Mediterranean and Black Sea rivers that drain catchments of $<5,000\ km^2$ are considered as small/medium rivers [23], hence the two rivers are considered as representative examples of large (Acheloos River) and medium/small (Louros) Mediterranean systems. Both river basins are located in the wet, western part of the country and receive annually 808 mm (Acheloos) and 925 mm (Louros) of precipitation [33, 34].

The Louros River discharges into the semi-enclosed Amvrakikos Gulf (Fig. 1), which is connected through a narrow, silled, natural channel to the Ionian Sea. The Amvrakikos wetland, consisting of the Louros and Arachthos deltas and several lagoons, is one of the most important protected wetlands of Europe, designated under the Ramsar Convention and the European Communities' Legislation.

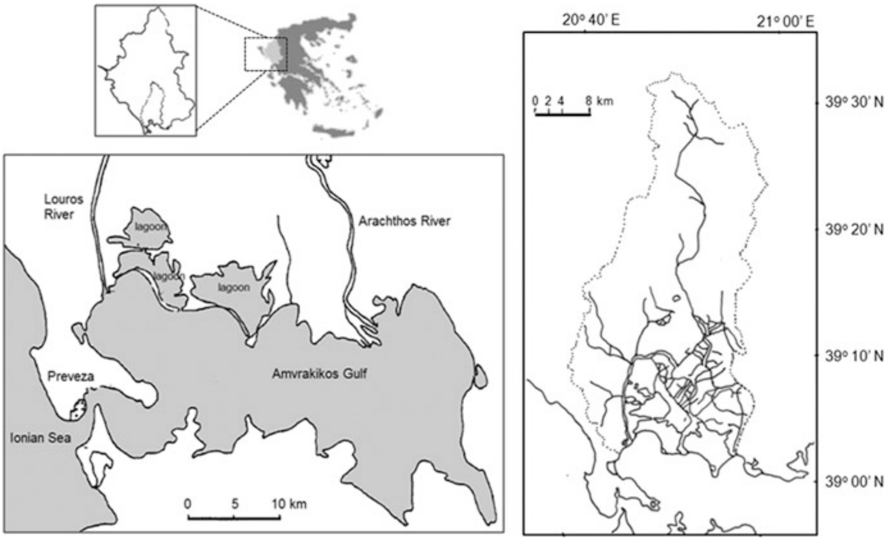


Fig. 1 The Louros watershed (marked with *dotted line*) and its estuary in the Amvrakikos Gulf. Note the dense network of artificial channels in the lower reaches of the watershed (*right panel*)

The Louros River has a length of approximately 80 km, an average width of 12 m, a depth of approximately 4.5 m and its mouth is silled by a very shallow bar of 0.6 m depth [28]. It has an average flow of 19 m^3/s [33]; however, extensive water abstraction for irrigation results in the decline of river runoff, particularly during the low flow season [28]. The drainage basin consists mainly of carbonate rocks (66%) and clastic (flusch and alluvial) sediments, resulting in relatively low sediment fluxes (0.8×10^6 t; [33]). Within the basin, more than 100 large and small agricultural industries are in operation [35].

The Louros Estuary provides insights onto the "microstructure" of a typical salt wedge system of low river flow and negligible tidal range. The highest stratification occurs during the summer months, when a deep pycnocline with considerable density gradients separates the fresh and saline water layers. During this season, saline water intrudes the estuary near the river bed, despite the existing shallow sill, and forms a thin (approximately 15 cm) salt–wedge water mass, which occupies the near-bottom layer with its thin end, pointed upstream. A detailed sampling took place during a period of minimum river runoff (10 m^3/s) and maximum penetration of saline water enhanced by southerly winds by employing a portable salinometer and a submersible micro-pump, which allowed for a thorough investigation at the fresh–saline water interface.

The second case study concerns the Acheloos River (Fig. 2), which is the second longest river in Greece (255 m). The Acheloos River is known from the Greek mythology as a river "God" fighting with Hercules over the river nymph Deianeira. The river is strongly fragmented by four large dams/hydroelectric plants located at the upper and middle part of the basin that have resulted in significant alteration of

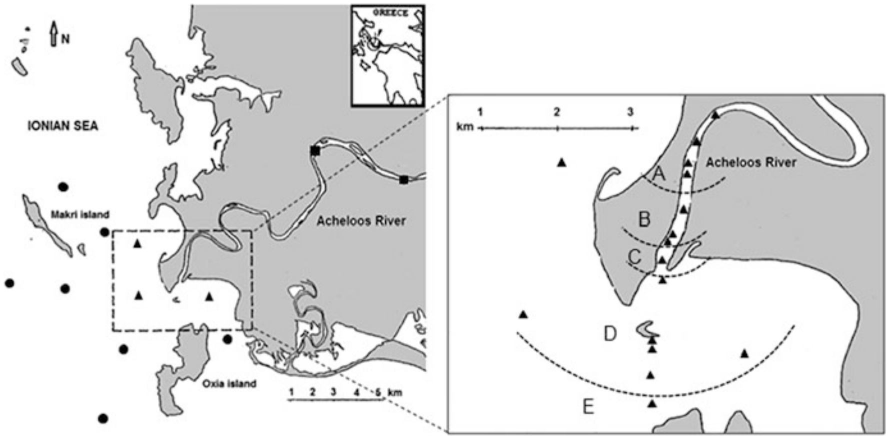


Fig. 2 The Acheloos River system (*right*), the Acheloos Estuary (*left*) and the grid of stations in the riverine (marked with *rectangles*), estuarine (marked with *triangles*) and the marine (marked with *circles*) sector of the system. *Left panel* shows the zones A–E (see text for description) of the estuarine sector

the hydrological regime as it concerns both the volume of the discharge and its seasonality: the highest maximum discharge occurs in July due to peak hydropower production. It is estimated that 30% of the annual flows occurs in short intervals during summer, compared to 11% prior to dam constructions [34]. The mean annual runoff at Kastraki (catchment area: 4,118 km^2) is 96.7 m^3/s and the range of measured monthly runoffs over the period 1980–2000 is between 13 and 1,118 m^3/s [34]. Runoff may decline further in some periods to less than 10 m^3/s, due to water abstraction for irrigation [25]. Furthermore, significant sediment retention in reservoirs (>80% of the annual sediment flux) has occurred with consequences in the sedimentation processes in the estuary.

At the river mouth there is a shallow, sand bar of 50–80 m width. The water depth at the bar area is less than 1 m, whereas upstream is approximately 4 m. The water depth increases abruptly (>40 m) at a distance of 3 km offshore [25, 32].

In the Acheloos Estuary seawater penetrates into the river bed and forms a salt wedge with significant salinity gradients between surface and bottom waters. During periods of limited freshwater supply, the fresh–saline water interface is shifted towards the upper part of the river, about 2–3 km from the river mouth. When the freshwater supply increases, the fresh–saline water interface obtains a vertical front, which lies close to the river mouth on the marine side of the bar [36]. The presence of a salt wedge of varying size and the seasonal variations of the high turbidity zone are some of the consequences of the significant river flow variations (in general reduction) due to the operations of the hydropower plants and other human interventions along the river course.

The Acheloos Estuary does not receive any direct industrial wastewater discharges from known point sources. However, it receives the land washout and agricultural discharges of a relatively large catchment area and cultivated lands near the estuary, where agricultural land covers 41% of the area [37]. The estuary is of high environmental importance, as the discharges of the Acheloos affect not only the important Messolongi lagoons and other smaller wetlands, but also the distributions of nutrients of the entire northwestern section of the Patraikos Gulf and the nearshore part of the Ionian Sea, acting as a significant nitrogen source, attributed largely to the washout of fertilisers [36, 38].

The Acheloos River has received international attention because of the ongoing plans for its diversion towards Thessaly and the Aegean Sea, for the past 30 years. The Acheloos Water Transfer Project includes the diversion of a large portion of the Acheloos waters (today at a reduced rate of 0.6 km^3/year) towards the Pinios River basin to serve hydropower generation, drinking water supply, irrigation and improvement of the surface and groundwater quality of the intensively cultivated Thessaly plain. This project has been blocked by various sectors of the Greek society including NGOs and several decisions of the High Courts [34, 39–41].

3 Materials and Methods

Physicochemical parameters, including salinity, pH and dissolved oxygen (DO), were measured in situ by portable instruments. The water samples were collected by mini, whole-plastic submersible pumps, horizontal Hydro-Bios and Niskin bottles depending on the water depth of each station. Surface sediments were collected by means of grab samplers and short cores, and by means of a pneumatic corer [42] and Perspex tubes.

In the laboratory, water samples were filtered through 0.45 μm Millipore filters. Trace metal in the dissolved phase was pre-concentrated on a Chelex-100 resin, following a slight modification of the Riley and Taylor [43] method after Scoullos and Dassenakis [44]. Water sample handling was carried out in a clean box to prevent contamination. In the case of the Louros case study, the filters holding the particulate matter were divided into two. One half of each filter was treated with boiling conc. HNO_3 in covered PTFE beakers, whereas the other half was treated with cold 0.5 N HCl [28]. In the case of the Acheloos River, the filters were digested only with boiling conc. HNO_3. In this review, the results of analyses of $n = 128$ water samples in the case of Acheloos Estuary and $n = 17$ water samples in the case of Louros Estuary are reported. The sediment samples of both the Louros (number of sediment samples $n = 12$) and Acheloos systems ($n = 14$) were digested in a hot plate with concentrated HNO_3. The labile, non-lattice held fraction of trace metals were extracted by the single-step 0.5 N HCl method of Agemian and Chau [45]. In addition to the dilute HCl extraction, the sequential extraction scheme (SES) described in Scoullos and Oldfield [46] and summarised in Table 1 was employed in the sediment samples of the Acheloos Estuary. Trace metal concentrations were determined by means of Graphite Furnace or Flame Atomic Absorption Spectroscopy.

Table 1 Reagents and leachable fractions of metals by sequential extraction scheme after Scoullos and Oldfield [46]

Extraction step	Reagents	Duration and temperature of extraction	Fraction of metals
1	1M $MgCl_2$	16 h, room temperature	Easily exchangeable
2	1M CH_3COOH-$NH_2OH \cdot HCl$	16 h, room temperature	Non-exchangeable, non-lattice held, inorganic
3	0.05M EDTA	24 h, room temperature	Organic
4	Conc. HNO_3: $HClO_4$ (2:1)	180°C evaporation until near dry (3 times)	Residual

In all studies, determinations of trace metal contents in sediments were performed in the <63 μm fraction in order to minimise grain-size effects [47]. As a second step, a geochemical normalisation to Al was applied, in order to reduce residual variance due to compositional (mineralogical) differences within and between the cores, despite the fact that the <63 μm fraction is considered as the least affected by grain-size effects [47, 48].

Organic carbon content was determined after the Gaudette et al. [49] method. Mineral magnetic measurements were determined by the methods and instrumentation described in Scoullos and Oldfield [46], and Scoullos, Botsou and Zeri [50].

4 Results and Discussion

4.1 The Louros Estuary

Physicochemical parameters (salinity, pH, Suspended Particulate Matter – SPM) allowed for the identification of three distinct water masses:

(a) The overflowing, seaward moving, riverine water mass. Salinity and pH range from 0.5 and 7.0 at the upper part of the river, respectively, to 4.2 and 7.3, respectively, at the lower reaches of the estuary.
(b) The salt water wedge, occupying the deep layer near the river bed. The salt wedge has a length of approximately 5.7 km, an average width of 15 cm, although these characteristics may fluctuate with time, depending directly or indirectly on seasonal phenomena, mainly the river runoff, the ambient temperature and the prevailing winds. The ranges of salinity and pH are 25.6–28.9 and 7.6–7.8, respectively.
(c) The marine water outside the river mouth, with typical marine water salinity (34.2–36.9) and pH values (7.8–8.2).

The aforementioned structure of the system is graphically presented in Fig. 3.

The descriptive statistics of Al, Fe and trace metals concentrations in the dissolved phase and their contents of the particulate matter (extracted by conc.

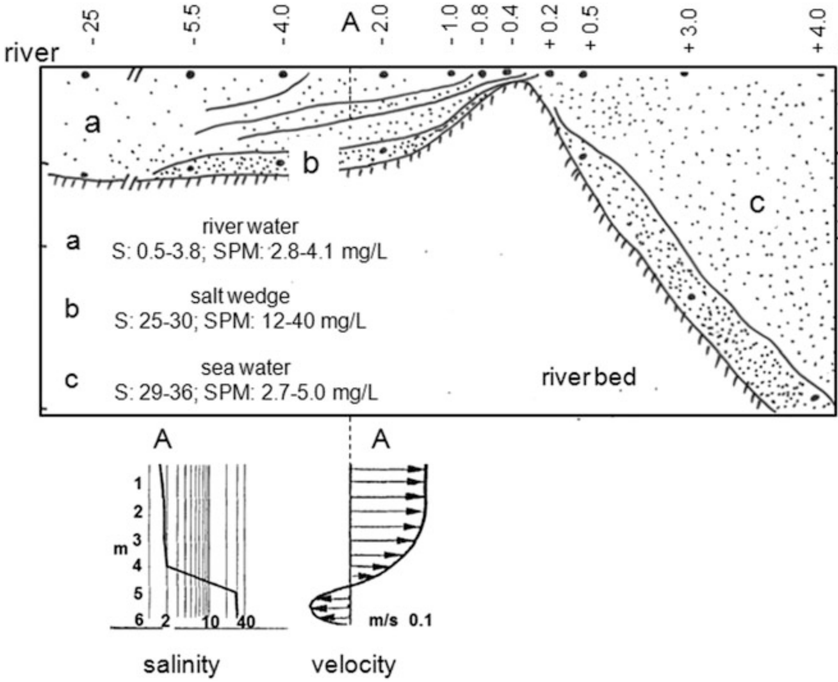


Fig. 3 Cross section of the Louros Estuary indicating the three distinct water masses: (**a**) the riverine, (**b**) the salt wedge and (**c**) the marine. Horizontal axis shows the distance of sampling stations (marked with *circles*) from the mouth of the river: (*minus*); upstream; (*plus*): offshore. Density of dots in the three water masses represents the concentration of SPM. Isohalines have been determined by in situ measurements. Velocity corresponds to average current. Modified from Scoullos et al. [28]

HNO_3) in the three masses of the system are given in Table 2. Dissolved Fe, Mn and Cu concentrations at the riverine water mass are much lower than the average concentrations of world's rivers [51], whereas dissolved Pb and Zn are slightly higher. Particulate Al and Fe are lower than the world's average values, because of the non-complete dissolution of the crystal lattice of aluminosilicate minerals by the boiling conc. HNO_3. Particulate Cu is lower, whereas particulate Mn, Pb and Zn are slightly higher than the world's average, indicating probably the influence of local or upstream anthropogenic activities.

With regard to the variations of dissolved trace metal concentrations in the three masses of the system, the data in Table 2 show that the trace metal levels in the marine water mass are similar (e.g. Pb), or even higher (e.g. Cu and Zn) than in the river water. This distribution pattern is rather unusual for most estuaries, where the adjacent seawater concentrations are normally lower. Furthermore, based on the average values, the highest concentrations of all the metals studied are found in the salt water wedge, indicating the addition of metals in solution at the salt stress interface (Table 2). Figure 4 shows the variation of dissolved trace metal

Table 2 Ranges and average (in parenthesis) concentrations of Al, Fe and trace metals in the dissolved (Diss.) and particulate (Part.) phase, as well as partition coefficient (K_D) in the three distinct water masses of the Louros system (number of samples, $n = 17$)

	Al	Fe		Mn		Cu		Ni		Pb		Zn	
	Part	Diss.	Part.	Diss.	Part.	Diss.	Part.	Diss.	Part.	Diss.	Part.	Diss.	Part.
RIV (a)	0.3–4.7 (3.8)	0.95–2.30 (1.39)	0.31–4.0 (3.0)	0.07–0.26 (0.15)	304–6,821 (5,145)	0.24–0.60 (0.38)	14.6–88.6 (58.6)	0.45–5.20 (1.60)	132–878 (454)	0.05–0.48 (0.24)	18.5–220 (71.8)	0.50–2.10 (1.33)	30.9–371 (272)
SW (b)	2.0–3.2 (2.7)	1.50–2.10 (1.83)	2.3–2.7 (2.5)	31.4–45.4 (36.4)	2,614–20,972 (10,528)	5.20–5.80 (5.43)	97.2–1,056 (524)	3.10–5.10 (4.00)	368–392 (383)	0.14–0.60 (0.30)	40–344 (166)	21.4–41.3 (28.1)	598–1,696 (1,102)
MAR (c)	0.3–2.3 (1.1)	1.00–2.00 (1.59)	0.3–1.5 (0.72)	0.08–0.46 (0.23)	391–1,471 (840)	0.40–0.85 (0.60)	19.4–59.4 (35.4)	0.90–2.40 (1.74)	111–361 (187)	0.15–0.30 (0.23)	8.2–62.5 (37.9)	0.82–2.05 (1.59)	50.8–269 (170)
ORM	1.98	2.70	1.1	0.50	500	2.20	28.6	9.20	127	0.50	29.4	2.30	50.8
World Rivers													
	8.7	*66.0*	*5.81*	*34.0*	*1,679*	*1.48*	*75.9*	*0.80*	*74.5*	*0.08*	*61.1*	*0.60*	*208*
K_D													
RIV (a)		6.13–7.62 (7.28)		6.07–7.98 (7.58)		4.39–5.48 (5.16)		4.40–6.20 (5.55)		4.59–6.64 (5.39)		4.31–5.81 (5.28)	
SW (b)		7.10–7.18 (7.13)		4.91–5.66 (5.33)		4.26–5.31 (4.81)		4.89–5.07 (4.99)		5.46–5.88 (5.70)		4.44–4.67 (4.58)	
MAR (c)		6.17–6.82 (6.56)		6.07–7.05 (6.54)		4.25–4.91 (4.67)		41.47–5.48 (4.89)		4.61–5.48 (5.10)		4.56–5.17 (4.95)	
ORM		6.75		6.47		4.25		4.47		4.77		4.57	
European estuaries		*Not given*		*Not given*		*4.0–5.0*		*4.0–4.5*		*5.0–7.0*		*4.0–5.0*	

Dissolved metal concentrations are expressed in μg/L. Particulate contents refer to the boiling conc. HNO_3 extractable fraction and are expressed in the (w/w) form (Al, Fe in %, trace metals in mg/kg). Data for the station which is located outside river mouth (ORM) and is affected by re-suspension are given separately. K_D is expressed in logarithmic values. For comparison we report also the average concentrations of major and trace elements in the dissolved phase and suspended sediments of World Rivers [51] and the K_D values reported in European estuarine and coastal systems, reviewed by Balls [52]

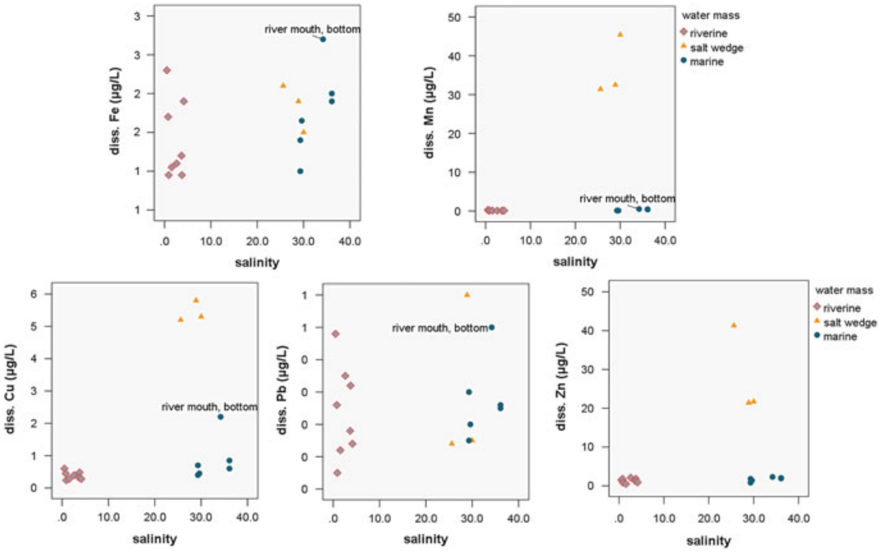


Fig. 4 Plots of dissolved trace metal concentrations against salinity in the Louros Estuary. The station outside the river mouth affected by re-suspension of sediments is marked as "river mouth, bottom"

concentrations with salinity. All metals show a non-conservative behaviour. Two different distribution patterns can be detected. The first distribution pattern is followed by Fe and Pb, for which the concentrations are highly scattered both at the low and the high salinity regime. The high variability of metal concentrations at the low salinity river water suggests the existence of local sources of these metals at the lower reaches of the river. Further downstream, the addition of dissolved metals is related to solid–solution interactions where the solid phase could be either the particulate matter, or re-suspended sediments, as in the case of the bottom water outside the river mouth (station shown in Fig. 4). Re-suspension of sediments at this station is evidenced by the high SPM concentrations, reaching 500 mg/L. The second distribution pattern is followed by Cu, Zn and Mn. In this case, the plots of dissolved metal concentrations against salinity show that addition processes clearly take place at the salt wedge.

Particulate metals (w/v), extracted by conc. HNO_3 and 0.5 N HCl, are accumulated in the salt wedge, as it becomes evident from their concentrations, which are considerably higher than those of water masses (**a**) and (**c**) (Figs. 5a–d). The enrichment of both leachable forms of metals in the salt wedge is mainly attributed to the high SPM concentrations in the water mass (**b**), with an average value (± 1 standard deviation) of 28.9 ± 14.4 mg/L, whereas in the river water it averages 8.88 ± 16.1 mg/L.

Figure 5e, g shows the variation of average values of 0.5 N HCl leachable trace metals contents (w/w) in the three masses of the system. This figure, combined with the data from Table 2 on trace metal contents of the particulate matter extracted by

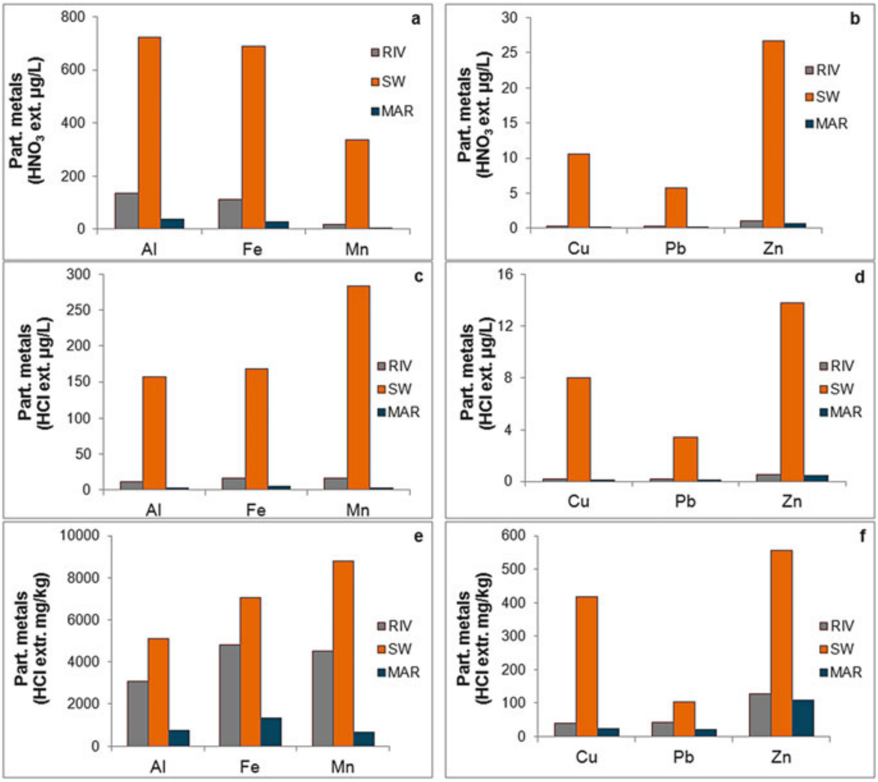


Fig. 5 Distribution of particulate metals in the riverine (RIV), the salt wedge (SW) and the marine (MAR) water masses of the Louros Estuary. (**a**, **b**): extracted by conc. HNO_3 (w/v); (**c**, **d**) extracted by 0.5 N HCl (w/v); (**e**, **f**) extracted by 0.5 N HCl (w/w)

conc. HNO_3, shows that for both fractions the particles of the marine water mass (**c**) have lower metal content than the riverine ones of water mass (**a**). Furthermore, Mn, Cu, Pb and Zn contents are the highest in the water mass (**b**) (Table 2). The parallel trend for the HNO_3 and HCl leachates of Mn, Cu and Zn is attributed to the fact that a considerable portion of the particulate metal is readily dissolved in dil. HCl; Mn extracted by the dil. HCl represents $83 \pm 9\%$ (mean $\pm$ sd) of Mn extracted by conc. HNO_3 ("pseudototal"); the HCl extractable fraction of particulate Cu and Zn represent $65 \pm 26\%$ and $64 \pm 18\%$, respectively, of the HNO_3 extracted metals.

Particulate Al and Fe contents extracted by conc. HNO_3 are reduced towards the marine sector of the system (Table 2). This seaward decreasing trend could be explained by mixing of fluvial, mineral particles with marine ones, which during summer, and low flow regimes are usually characterised by an increased contribution of the planktonic component in the overall suspended load [7, 9]. However, when considering the 0.5 N HCl leachable Al, Fe and Mn contents (Fig. 5e) it becomes evident that the highest values are found in the particles of water mass (**b**). This finding indicates the addition of Al, Fe and Mn forms that are easily extracted by dil. HCl in the salt wedge. At the "salt-stress" surface, which separates water mass (**a**) from (**b**), the low pH Louros water meets the saline water of relatively higher pH. Formation of authigenic particles, especially Fe and Mn oxyhydroxides, is known to take place in similar interfaces [4, 15, 53], resulting in the enrichment of these metals in the particles of the salt wedge. In water mass (**b**), the extractability of Al by the 0.5 N HCl increases to 20 $\pm$ 7% compared to 8 $\pm$ 5% and 6 $\pm$ 3% in water masses (**a**) and (**c**), respectively. Thus, the observed Al-enrichment of the HCl-soluble fraction in water mass (**b**) could be related to the precipitation of Al-oxyhydroxides, or most probably to coagulation of fine, poorly crystalline aluminosilicates under the influence of increased ionic strength.

In order to gain insights into the solid-dissolved phase interactions, the partition coefficient K_D is employed, which is defined as follows: K_D = Particulate metal concentration (w/w)/Dissolved metal concentration (w/v) [16]. Summary statistics of K_D in the three water masses of the system is shown in Table 2.

Average K_D for each element decreases in the order Fe > Mn > Pb > Zn > Cu, indicating the affinity of Fe and Mn for the solid phases, whereas the lower K_D for Cu and Zn indicates their affinity for the dissolved phase. Similar K_D values are reported in other European Estuaries (Table 2), as reviewed by Balls [52]. The K_D values for Fe, Cu and Pb gradually decrease from the riverine water mass to the salt wedge and then to the marine water mass, whereas the K_D for Mn and Zn substantially decreases at the salt wedge in relation to the river water and further increases at the marine water mass. Characteristic plots K_D for Fe and Mn with salinity are shown in Fig. 6a, b.

The highest K_D values for Fe are found at the riverine water mass. Although coagulation of colloidal Fe could be significant at salinities of <5‰ [54], as dissolved Fe does not follow any clear trend with increasing salinity, addition of Fe from local sources at the low reaches of the Louros Estuary seems to be responsible for the elevated K_D values at this part of the system. Further downstream, the K_D for Fe decreases compared to the riverine water mass. This is despite the precipitation of Fe oxyhydroxides at the salt wedge, which was discussed previously. It should be noted that the particulate metal contents extracted by the boiling conc. HNO_3 are used for the calculation of K_D. Thus, the authigenic component of Fe in the pseudototal content is masked by the increased contribution of other mineral Fe-bearing phases, such as clays and crystalline oxyhydroxides. Apparently, addition of Fe in solution takes place at the salt wedge that is responsible for the decreased K_D values at this part of the system.

Sediment–water interactions and benthic fluxes could be an important source of dissolved metals, particularly in (periodically) anoxic or suboxic systems [5, 18, 55, 56]. The onset of anaerobic conditions due to initial oxygen consumption during organic matter decomposition results in the formation of metal authigenic sulfides and/or carbonate phases. These phases are stable under anaerobic conditions. There are two mechanisms responsible for the benthic fluxes of metals [55]: Firstly, metals may be released by molecular diffusion from pore waters. The diffusion is enhanced by higher temperatures and/or formation of dissolved metal species (e.g. Fe(II), Mn(II)) by reductive dissolution of Fe/Mn oxyhydroxides. In this way,

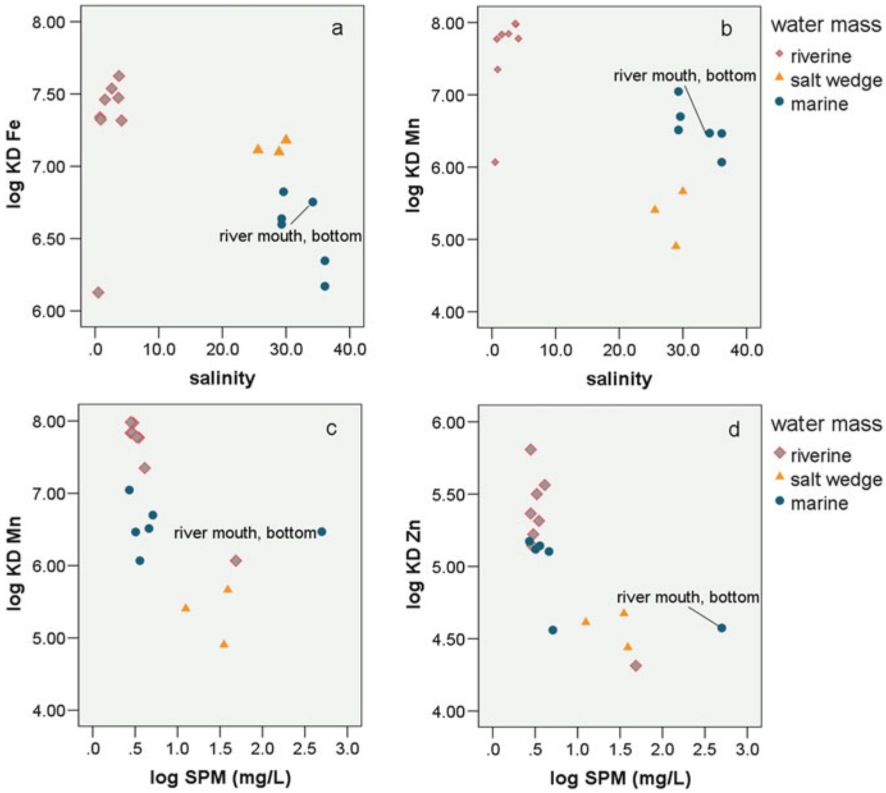


Fig. 6 Plots of partition coefficients (K_D) for Fe (**a**) and Mn (**b**) against salinity and K_D for Mn (**c**) and Zn (**d**) against suspended particulate matter (SPM)

trace metals previously bound to Fe/Mn oxyhydroxides could also be transported to the dissolved phase. Secondly, metal fluxes can be enhanced by physical, biological disturbances of sediments, i.e. re-suspension during flooding events and/or bioturbation, or salt wedge migration. In this case, apart from direct injection of pore waters into the water column, desorption from the suspended particles, oxidative dissolution of reduced authigenic solid phases (e.g. sulfides) could result in increased dissolved metal concentrations.

The observed further decrease of K_D for Fe (as well as Cu and Pb) at the marine water mass is attributed to the dilution of enriched riverine particles with marine, metal-poor particles of biogenic origin. This is supported by the decreased metal contents of suspended particles (in the w/w expression). Additionally, re-suspension of sediments, followed by emanation of dissolved metal species from pore waters into the water column and most probably desorption of metals from the re-suspended sediments, might also be responsible for high dissolved metal concentrations and the decrease of K_D for these metals. Re-suspension of sediments evidenced by the high SPM concentrations (500 mg/L) and subsequent effects on the partitioning of Fe and trace metals between the solid phase and solution is best distinguished at the bottom water outside the river mouth (see Figs. 4 and 6a).

Similar processes could account for the variability of K_D for Cu and Pb along the three distinct water masses of the system. The increased dissolved metal concentrations observed in the salt wedge (Fig. 4) result in the decrease of K_D for these metals. Apart from the sediment–water interactions, previously described for Fe, desorption from suspended particles is an important mechanism releasing trace metals into the bottom, salt water mass. In this part of the system, the percentages of easily (0.5 N HCl) extractable particulate metals in relation to the pseudototal (HNO_3 extractable) content decrease compared to the riverine water, from 73% to 62% for Cu and from 66% to 62% for Pb. This finding suggests that weakly bound metals to suspended particles are released in solution. Desorption of metals is enhanced by the entrapment of suspended particles at the salt wedge, evidenced by the increased concentrations of SPM, and the long residence time of suspended particles during periods of low flow regime.

The partition coefficient K_D for Mn and Zn show an inverse relationship with SPM (Spearman correlation coefficients $r = -0.737$; $p = 0.001$ and $r = -0.771$; $p < 0.0005$, respectively), which is graphically illustrated in Fig. 6b. This relationship, known as particle concentration effect, has been observed in many estuaries (e.g. [2, 7, 57]). Several causes have been proposed for the decline of K_D with the increase of SPM concentrations, including the abundance of coarse particulate matter in occasions of high SPM load, which have lower surface area, hence fewer complexation/sorption sites per mass, the pronounced removal of trace metals in the estuarine turbidity maximum [58], but most often the existence of colloidal particles of <0.45 μm size that are counted with the dissolved fraction [59]. With increasing concentrations of SPM, the concentration of fine-sized colloids increases. These colloidal particles are able to bind metals and retain them in solution, hence the "dissolved" metal concentrations increase and K_D values decrease [7].

Part of the fine-sized colloidal particles could be removed from solution after coagulation under the influence of increased ionic strength. By conducting laboratory mixing experiments Sholkovitz [60] showed that the removal of Mn levels at salinities between 15 and 25‰, and additional removal occurs at salinities between 27 and 30‰. Thus, removal of Mn (and Zn) from the solution at the high salinity, marine water mass, combined with re-suspension of sediments, likely explain the observed increase of K_D for these metals from the salt wedge to the marine waters.

The surface sediments of the Louros Estuary are fine grained, with the mud fraction representing more than 95% of the entire sediment throughout the system. The organic carbon content increases from 0.6% in the upper part of the river to approximately 2.3% at the intermixing zone. The carbonate content does not fluctuate significantly along the system and ranges from 28% to 33%.

Table 3 summarises the results of extractions by conc. HNO_3 and 0.5 N HCl of surface sediment samples of the Louros Estuary. Compared to the composition of upper continental crust (UCC; [61]), Cu and Zn contents of the Louros surface

Table 3 Ranges and average (in parenthesis) of Al, Fe and trace metal contents in the surface sediments at the riverine and marine sector of the Louros system (number of samples, $n = 12$)

	Al		Fe		Mn		Cu		Ni		Pb		Zn	
	HCl	HNO_3	HCl	HNO_3	HCl	HNO_3	HCl	HNO_3	HCl	HNO_3	HCl	HNO_3	HCl	HNO_3
RIV (a)	0.25–0.37 (0.29)	1.42–5.05 (3.05)	0.45–1.00 (0.72)	1.35–4.05 (2.29)	350–760 (511)	390–870 (618)	18.5–38.3 (29.0)	28.8–51.9 (38.6)	42.0–87.8 (61.8)	113–242 (178)	5.50–11.5 (7.88)	6.10–13.0 (9.42)	24.2–79.2 (40.3)	36.1–107 (79.9)
SW (b)	0.28–0.35 (0.33)	3.65–5.05 (4.30)	0.84–1.00 (0.95)	2.24–4.05 (3.10)	510–760 (573)	610–810 (733)	31.0–38.3 (33.8)	42.5–51.9 (47.1)	42–64.6 (63.2)	183–242 (214)	7.80–8.60 (9.30)	10.6–13.0 (11.9)	24.2–79.2 (43.6)	89.4–107 (95.5)
MAR (c)	0.24–0.31 (0.27)	1.84–3.19 (2.55)	0.53–0.75 (0.53)	1.45–3.25 (2.47)	350–380 (370)	420–640 (513)	20.0–25.8 (23.4)	26.0–38.9 (33.3)	40.4–69.2 (51.1)	116–200 (168)	4.10–10.1 (7.33)	7.20–11.8 (9.27)	26.6–40.8 (31.6)	75.3–90.6 (82.3)
UCC	*8.15*		*3.5*		*774*		*28*		*47*		*17*		*67*	

The riverine sector includes stations located landwards of the sill bar and those overlain by the salt wedge development at the specific time of sampling, whereas the marine sector includes stations located seawards of the sill bar. For comparison with the composition of SPM, sediment samples of the salt wedge at which saline bottom waters were sampled are also given separately (b). Sediment samples were extracted by 0.5 N HCl (HCl) and boiling conc. HNO_3 (HNO_3). Al and Fe contents are given in % (w/w), whereas trace metal contents in mg/kg. For comparison we report also the average metal contents of the upper continental crust (UCC) reported by Rudnick and Gao [61] sediments are slightly higher than the UCC, whereas Pb levels are lower than the UCC values.

Sediments of the riverine sector contain higher amounts of both fractions of metals than those of the marine sector. The percentages of Al extracted by the diluted HCl in relation to the conc. HNO_3 are stable across the system and have an average value of 11%, indicating common sources of Al throughout the estuary. This observation, combined with the fact that Al contents (HNO_3 extractable) decrease from 1.42–5.05% to 1.84–3.19% seawards, signifies that part of the fine-sized aluminosilicates are entrapped at the upper estuary due to presence of the sill bar (Fig. 3). The same pattern is observed for Fe and Mn as well. The extractability of Fe and Mn decreases from 34% and 84% at the riverine sector to 22% and 74% at the marine sector, respectively. This suggests that Fe/Mn oxyhydroxides (easily extractable with both acids), part of which are authigenically formed at the fresh–saline water interface, are also entrapped landwards from the sill bar. As clays and Fe/Mn oxyhydroxides are efficient scavengers for trace metals [47], the physical entrapment of these phases could also explain the relative enrichment of trace metals at the riverine sediments.

Sediments obtained from the stations affected by the active salt wedge at the time of sampling contain higher amounts of Al and Fe, but much lower amounts of metals compared to the overlying suspended matter (see metal contents of the SW sector in Tables 2 and 3). Apparently, the enrichment of the suspended particles during low flow conditions is reflected only to a small degree at the surface sediments that represent a long-term and variable flow regimes repository.

After normalising the metal contents to Al, in order to compensate for grain-size effects, a different distribution pattern of trace metals emerges. Figure 7 shows that higher metal to Al ratios are observed at the outer, marine sector of the Louros system. Thus, it is proposed that particles enriched in trace metals during their entrapment in the salt wedge by the processes described previously are transported and eventually accumulated seawards, during high flow regimes and flooding events.

Figure 8 illustrates representative vertical distributions of the HNO_3 extractable fraction of Cu in short cores obtained from the intermixing zone and an offshore station. It is clear that, despite the variability of the metal content at the various sites sampled, there is a general increase of Cu content towards the younger, surface sediments. This trend, which is also followed by Zn, Cr and Pb [28], suggests a gradual increase of pollution from a combination of small point and non-point sources.

Previous detailed mineral magnetic studies at the Louros Estuary [46], showed that the upper part of the cored sediments is dominated by fine grained, soil-derived material, rich in secondary spinel oxides, whereas the lower part of the cores has a much harder demagnetisation behaviour, which is attributed to the presence of hematite-coated sand grains clearly of detrital origin. The latter magnetic component has a different provenance, or derives from an earlier denudational regime. According to the study of Scoullos and Oldfield [46], Cu, among other metals, correlates with volume-specific magnetic susceptibility *k* and frequency dependent

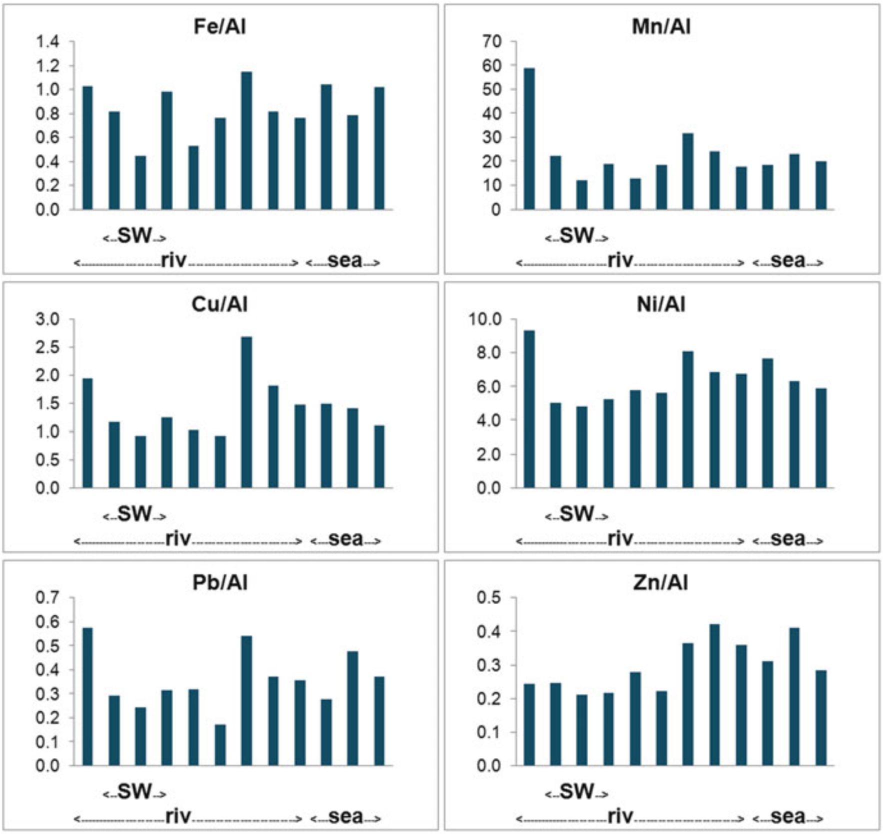


Fig. 7 Distribution of metal contents, extracted by boiling conc. HNO_3, normalised to Al (metal/Al) in the surface sediments of the Louros Estuary

magnetic susceptibility kfd%, only in one of the major components of the system, namely the fine particles deriving from soil erosion, predominantly found at the upper part of the cores. It is well known that the fine fraction of particles is the most important one for the transport of metals [47]. It is, thus, concluded that the enrichment of the surface layers of the cores is attributed to the transport of metals through land washout and runoff, with some contribution of the re-precipitation of dissolved metals occurring at the upper layers of the cores.

Summarising the results of the Louros system, a mechanism of metal enrichment within the intermixing zone could be proposed: After the separation of the heavier particles by precipitation at the upper part of the river, the lighter ones, with small grain sizes and higher content of metals, organic matter and minerals, remain in suspension until they reach the lower part of the estuary. A large proportion of these particles are accumulated at the stable fresh–saline water interface and at the thin intermixing zone and is then trapped and recycled within the landward moving saline layer, along the river bed (the salt wedge). Desorption of metals from mineral

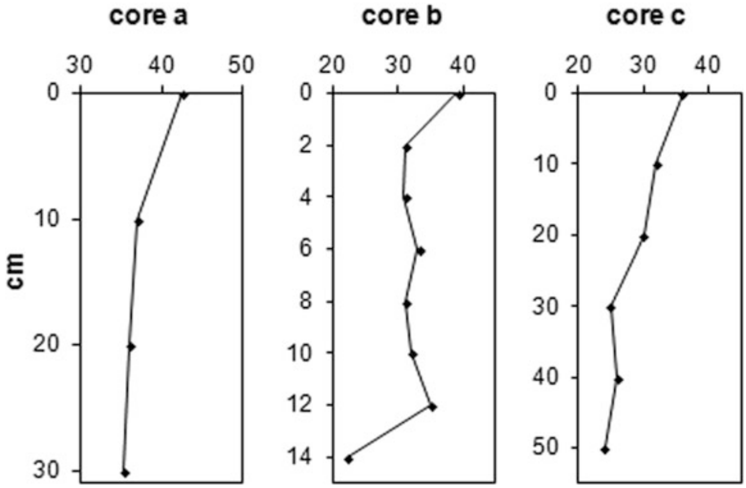


Fig. 8 Vertical distribution of Cu contents (in mg/kg) extracted by boiling conc. HNO_3 in cored sediments obtained from the estuarine (cores **a**, **b**) and the marine sector (core **c**) of the Louros River

surfaces of suspended particles (SPM) and/or re-suspended sediments, as well as benthic fluxes driven by diagenetic, redox processes in sediments, which provide the observed increases of dissolved metal concentrations in the salt wedge, is paralleled by an almost simultaneous formation of authigenic particles and colloidal, most probably Fe/Mn oxyhydroxides and Al clays. The metals trapped into the salt wedge can be released in the Amvrakikos Gulf when the river flow increases and the Louros Estuary becomes flooded.

4.2 The Acheloos Estuary

Based on the physicochemical and hydrological conditions, water samples obtained from the river course of Acheloos are divided into four types:

(a) The overflowing, seaward moving, riverine water mass, which includes samples of salinity <3 and water depth 0–0.5 m.
(b) The estuarine water mass, which includes samples of salinity 6.4–33.6 and water depths 0–5 m.
(c) The salt wedge, occupying the subsurface layer near the river bed, which includes samples of salinity 11.4–37.8 and water depths ranging from 1 to 5 m.
(d) The marine water outside the river mouth, which includes samples of salinity 36.1–38.1 and water depth 0–40 m.

Table 4 summarises the ranges and average values of Al, Fe and trace metals in the dissolved and particulate (extracted by conc. HNO_3 in w/w) phase in the

Table 4 Ranges and average (in parenthesis) concentrations of Al, Fe and trace metals in the dissolved (Diss.) and particulate (Part.) phase, as well as partition coefficient (K_D) in the four distinct water masses of the Acheloos system ($n = 128$)

	Al	Fe	Mn		Cu		Pb		Zn	
	Part	Part.	Diss.	Part.	Diss.	Part.	Diss.	Part.	Diss.	Part.
RIV (a)	0.15–6.46 (3.66)	0.10–4.32 (2.28)	0.03–2.75 (0.49)	797–6,283 (2,202)	0.01–5.40 (0.84)	4.2–110 (45.4)	0.07–2.85 (0.47	8.1–130 (39.5)	0.28–21.7 (5.88)	126–2,609 (576)
EST (b)	0.54–5.25 (2.88)	0.49–3.91 (2.12)	0.04–2.50 (0.26)	333–4,049 (1,123)	0.20–2.99 (0.73)	9.9–123 (55.5)	0.06–2.23 (0.35)	9.8–721 (76.2)	0.40–35.1 (6.27)	67–4,019 (815)
SW (c)	0.63–3.10 (1.93)	0.24–3.35 (1.93)	0.04–2.69 (0.58)	259–4,152 (1,997)	0.03–0.90 (0.23)	3.9–104 (35.8)	0.11–4.13 (0.39)	7.7–814 (123)	3.90–21.8 (12.1)	180–2,771 (636)
MAR (d)	0.05–2.05 (0.54)	0.06–2.09 (0.46)	0.03–0.80 (0.25)	21–2,000 (406)	0.11–2.83 (0.62)	3.8–471 (106)	0.11–4.13 (0.39)	9.6–580 (91.1)	0.55–46.5 (3.53)	50–6,170 (1,013)
K_D										
RIV (a)			5.84–7.76 (7.03)		3.15–6.29 (4.98)		3.64–5.80 (4.99)		4.31–6.04 (5.16)	
EST (b)			6.03–7.53 (6.84)		3.84–5.52 (4.90)		4.61–6.04 (5.26)		4.08–6.73 (5.09)	
SW (c)			5.02–7.84 (6.91)		3.96–6.42 (5.20)		4.29–6.14 (5.01)		4.12–5.45 (4.59)	
MAR (d)			4.93–7.24 (6.05)		3.84–6.53 (5.16)		3.83–6.08 (5.20)		4.10–6.47 (5.30)	

Dissolved metal concentrations are expressed in μg/L. Particulate metal contents are expressed in % for Al and Fe and in mg/kg for trace metals. K_D is expressed in logarithmic values. The marine sector includes samples obtained from the surface and from 10, 20 and 40 m depths aforementioned water masses of the Acheloos system. Dissolved Mn and Cu concentrations are lower than the average values of World's Rivers reported by Viers et al. [51] (Table 2), whereas Pb and Zn are higher. Average particulate metal contents are lower than the world's average values, except for Zn. The Acheloos River receives land runoff from a large catchment area, influenced by agricultural activities and atmospheric depositions, and probably urban effluents from small cities. Nevertheless, compared to other large perennial Greek rivers, the Acheloos system could be characterised as relatively unpolluted [32].

Considering the variations of dissolved metal concentrations in the four water masses of the system, on average, the highest concentrations of metals, except Cu, are determined in the salt wedge (Table 4). The addition of Mn and Pb in solution at the salt stress interface is related to interactions between the dissolved phase and suspended particles and/or re-suspended sediments. Based on the average values, dissolved Cu decreases from the riverine to the estuarine water mass and the salt wedge and further increases at the marine waters. Occasionally, deviations from these general distribution patterns are observed, when distinct sampling cruises are considered. Figure 9a shows the plots of dissolved metal concentrations with salinity during one month in the summer. In this case, dissolved Cu, Pb and Zn concentrations increase with increasing salinity in the estuarine zone, either gradually (Cu, Pb) or locally (Zn). Dissolved Mn concentrations do not vary widely between the riverine and the estuarine water mass. Nevertheless, the most interesting feature is probably the increased dissolved Mn, Cu and Pb concentrations in the marine waters, indicating that release processes occur not only in the intermixing zone, but also in the outer, marine stations of the system.

Figure 10 shows the distribution of average SPM and particulate metal concentrations in the four water masses of the system, as well as the plots of SPM, and particulate metals (in the w/v expression) with salinity during one month in the summer. On average, SPM concentrations decrease seawards due to dilution and deposition of coarse-grained fluvial particles before, or around the sill bar at the river's mouth [32]. High concentrations are determined at the bottom water of the salt wedge, due to re-suspension of sediments and trapping of settling particles, which is clearly depicted when a single sampling cruise is considered.

Average particulate Al, Fe and Mn concentrations (in the w/v expression) follow the distribution of SPM (Fig. 10a). They vary widely in the river water, in response to the hydrological regime, decrease in the estuarine mass but become enriched in the salt wedge, and further decrease in the marine waters. Particulate Cu and Pb are enriched in the estuarine waters and the salt wedge. These general distribution patterns are observed when a single sampling is considered, too (Fig. 10b). Particulate Zn does not vary greatly between the four masses of the system. During the summer month, the highest Zn concentration is determined at a marine station, probably due to the uptake of this element by phytoplankton.

Particulate Al and Fe contents (in the w/w expression) decrease gradually seawards, due to dilution of fluvial particles with biogenic, Al/Fe- poor, marine particles (Table 5). Particulate Mn exhibits its highest values in the riverine water mass and the salt wedge. The enrichment of particulate Mn probably indicates the

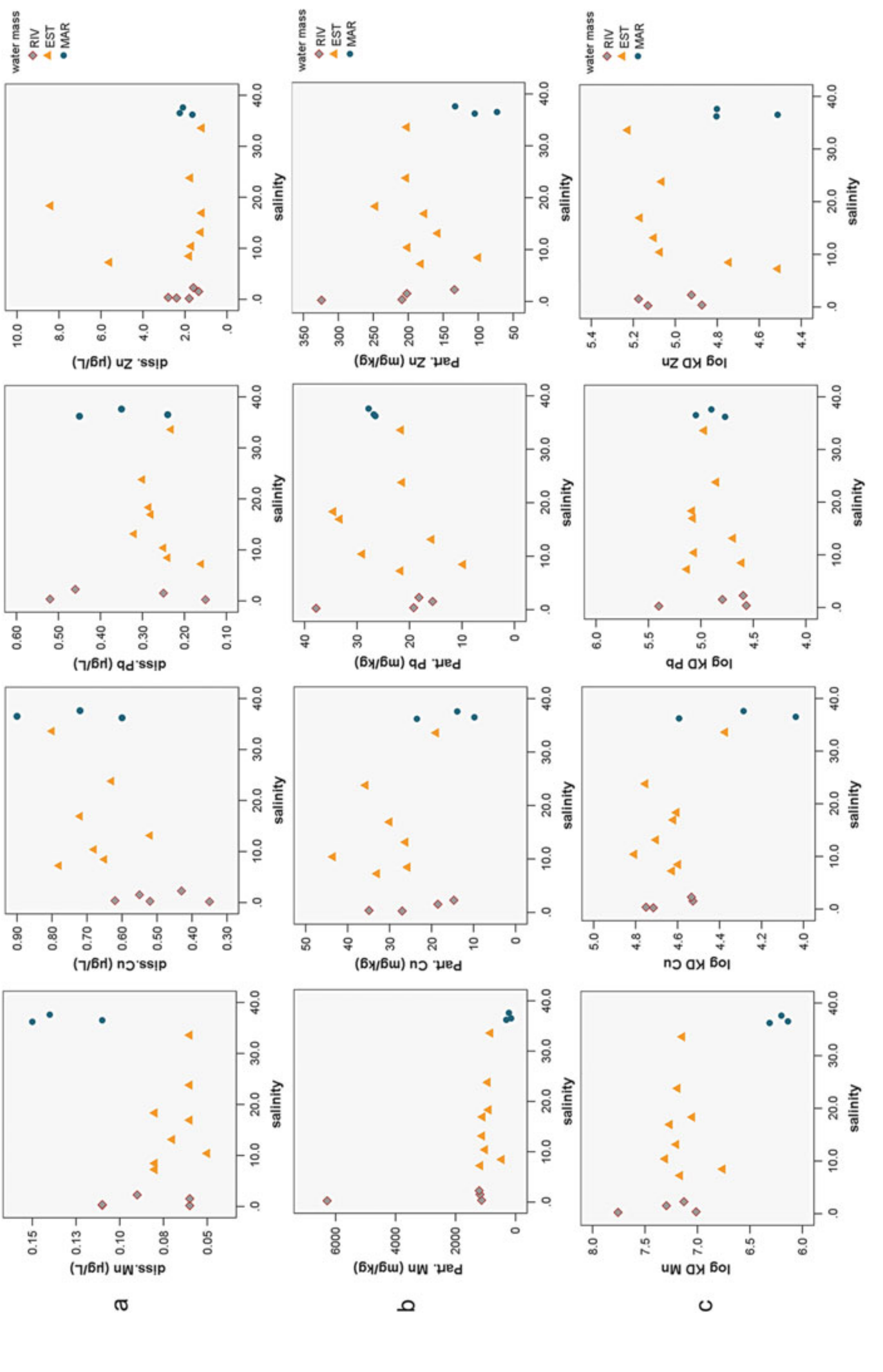


Fig. 9 Plots of (**a**) dissolved metal concentrations; (**b**) particulate metal contents (w/w); (**c**) partition coefficients K_D for Mn, Cu, Pb and Zn with salinity in the Acheloos system during a summer month

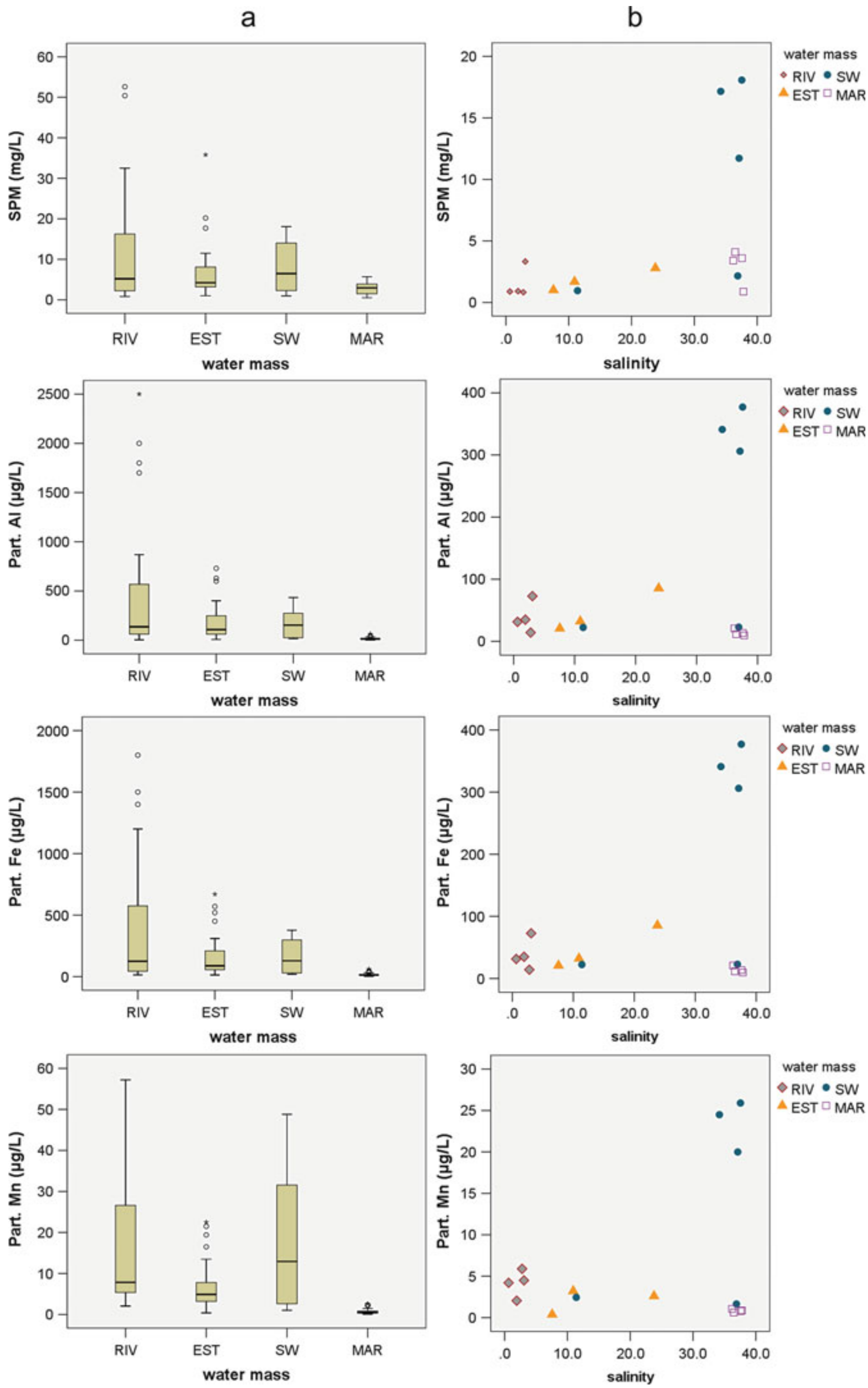


Fig. 10 Distribution of SPM and particulate metal concentrations (w/v) in the four water masses of the Acheloos Estuary: (**a**) Boxplots in the *left panel* summarise statistic data of all samplings; (**b**) scatter plots in the *right panel* show the variations of the studied parameters with salinity during a summer sampling cruise

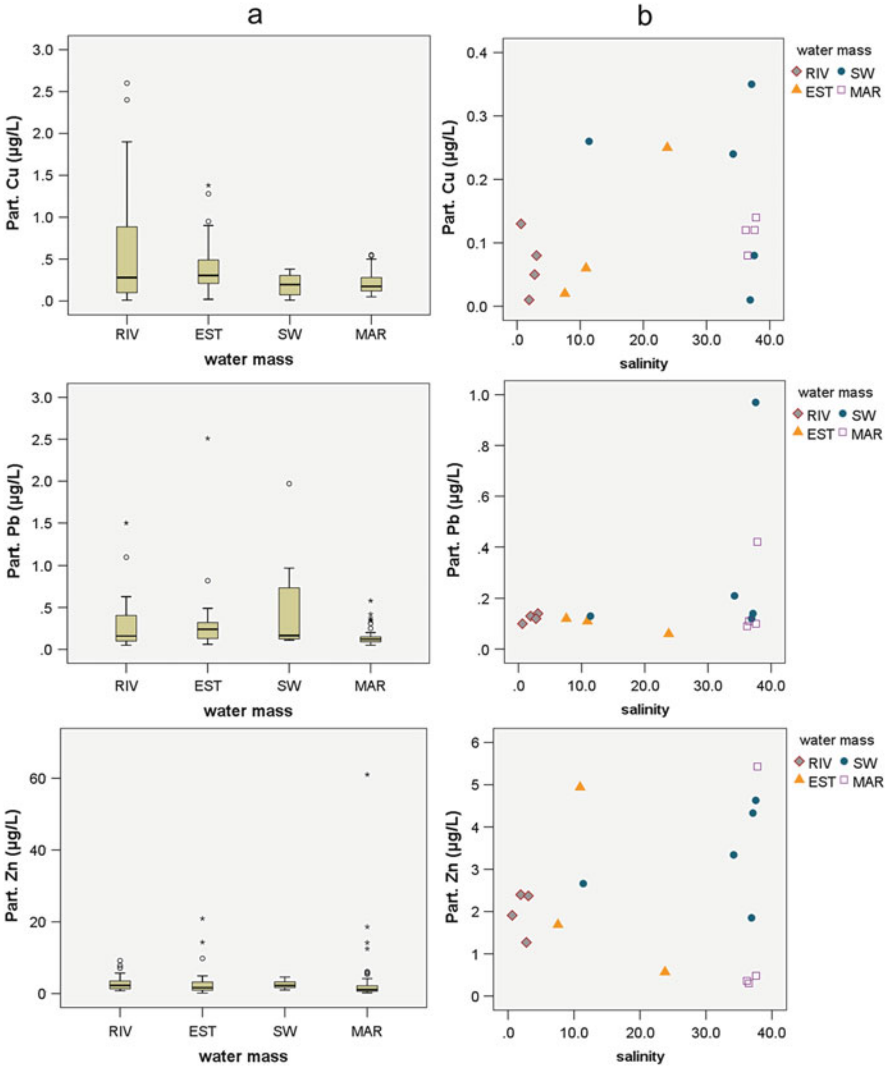


Fig. 10 (continued)

formation of authigenic Mn oxyhydroxides [32]. Particulate Cu, Pb and Zn are enriched, either in the estuarine water mass or the salt wedge. Scavenging by Mn oxyhydroxides or re-suspension of sediments are the likely sources of Cu, Pb and Zn enriched particles.

Figure 9c shows the variation of partition coefficient (K_D) for Mn, Cu, Pb and Zn as a function of salinity during a summer cruise. By combining the distribution patterns of K_D with those of dissolved metal concentrations (Fig. 9a) and particulate metal contents (Fig. 9b), some interesting observations on solid–solution interactions might be deduced.

Table 5 Ranges and average (in parenthesis) of Al, Fe and trace metal contents of the surface sediments at the five zones of the Acheloos Estuary (number of samples, $n = 14$)

	Al		Fe		Mn		Cu		Pb		Zn	
Zone	HCl	T	HCl	T	HCl	T	HCl	T	HCl	T	HCl	T
A	0.12–0.30 (0.23)	2.08–2.98 (2.41)	0.13–0.66 (0.36)	1.28–1.80 (1.57)	410–650 (550)	500–660 (597)	21.8–24.3 (23.1)	25.5–31.3 (28.8)	6.2–9.9 (7.6)	7.6–12.4 (9.9)	16.4–24.0 (19.8)	44.2–57.6 (49.7)
B	0.02–0.06 (0.05)	0.56–0.86 (0.74)	0.14–0.24 (0.20)	0.52–0.89 (0.74)	400–500 (463)	480–560 (520)	4.20–11.0 (7.53)	10.1–13.0 (11.6)	2.5–3.6 (3.0)	5.8–7.0 (6.2)	8.6–15.9 (11.6)	28.8–43.4 (37.0)
C	(0.28)	(2.80)	(0.41)	(1.91)	(380)	(470)	(26.2)	(38.4)	(9.7)	(11.3)	(25.6)	(81.7)
D	0.04–0.06 (0.05)	0.45–0.71 (0.53)	0.09–0.18 (0.14)	(0.24–0.49) (0.38	520–680 (625)	560–730 (663)	3.70–11.6 (5.90)	7.0–12.9 (9.4)	2.4–3.3 (2.8)	3.7–6.2 (5.0)	7.2–22.2 (15.3)	24.8–45.3 (32.5)
E	(0.28)	(3.96)	(0.45)	(2.50)	(470)	(510)	(20.5)	(32.3)	(14.4)	(15.1)	(34.9)	(72.8)

Zone A represents the upper estuary where the upper boundary of the salt wedge is observed during the low flow seasons; zone B extends over the area where the salt wedge is observed most of the time; zone C consists of a single station located at the river mouth and the sill bar; zone D is the lower estuary, off the river's mouth; zone E consist of one offshore station of the lower estuary. Sediment samples were extracted by 0.5 N HCl and conc. HNO_3:$HClO_4$ (T). Aluminium and Fe contents are given in % (w/w), whereas trace metal contents in mg/kg

Similar to the Louros Estuary, dissolved Mn increases at the low salinity (<10) regime. At the riverine and the estuarine water masses the partition coefficient K_D for Mn is inversely correlated to SPM (Spearman correlation coefficient $r = -0.556$; $p = 0.046$), indicating that the increased dissolved Mn concentrations correspond to increased colloids rather than truly dissolved forms (particle concentration effects). Colloidal Mn oxyhydroxides could be transferred seawards, or deposited at the estuarine zone after coagulation, depending on the river's flow regime. According to Dassenakis, Scoullos and Gaitis [32] these forms, together with Mn and Fe oxyhydroxides coating small particles and clays, are the main carriers for trace metals in the Acheloos system. The K_D for Mn decreases sharply at the outer marine stations. This decline is accounted for by the combined effect of the elevated dissolved Mn concentrations at the marine stations and the reduction of particulate Mn contents. Therefore, it is suggested that desorption processes and dilution of fluvial particles with the marine ones are taking place almost simultaneously. In the case of Cu, the increase of K_D at the mid-salinity regime is attributed to the increased particulate Cu contents, which likely ascribe to re-suspension of sediments. This is despite the increase of dissolved Cu concentration in the same region. Probably, the addition of Cu enriched particles from the sediments masks desorption processes. Similar processes could explain the variations of K_D for Pb, and Zn at the low- and mid-salinity regimes. Nevertheless, the most remarkable observation for all the elements presented in Fig. 9 is the elevated dissolved concentrations at the marine stations, which in combination with the decrease of K_D values, signify the importance of desorption processes at this part of the system.

At the marine stations, water samples were obtained from the surface, from 20 m and 40 m depth of the water column. The variations of SPM, particulate Al, dissolved and particulate Pb, as well as its partition coefficient with depth during a spring and a summer month (the same as in Fig. 9) are shown in Fig. 11a, b, respectively. The concentration of suspended particles varies with depth and increases either at both subsurface layers (spring; Fig. 11a) or at 40 m (summer; Fig. 11b). Particulate Al does not follow the distribution of SPM, suggesting the increased contribution of biogenic particles to the overall suspended load. Re-suspension of sediments could be an additional source of particles at the near-bottom waters, particularly during spring, when increased SPM concentrations are accompanied by increased Al contents at 40 m depth. Dissolved Pb concentrations are higher in the subsurface waters, than in the surface. At 40 m depth, oxic conditions prevailed during all samplings, with dissolved oxygen saturation ranging from 80% to 110% [32]. Thus, the increasing concentrations could not ascribe to benthic fluxes due to redox reactions. The K_D for Pb shifts to lower values with increasing depth of the water column, whereas particulate Pb follows the reverse trend. Release processes to the solution, which are evidenced by the decrease of K_D from the surface to the subsurface waters, suggest that desorption processes are taking place at the marine water column, due to competing and complexing processes with seawater ions. Increasing concentrations in solution with increasing depth of the water column are observed for Mn, Cu and Zn and desorption processes are recognised for these elements, too [32]. The abundance of organic ligands in

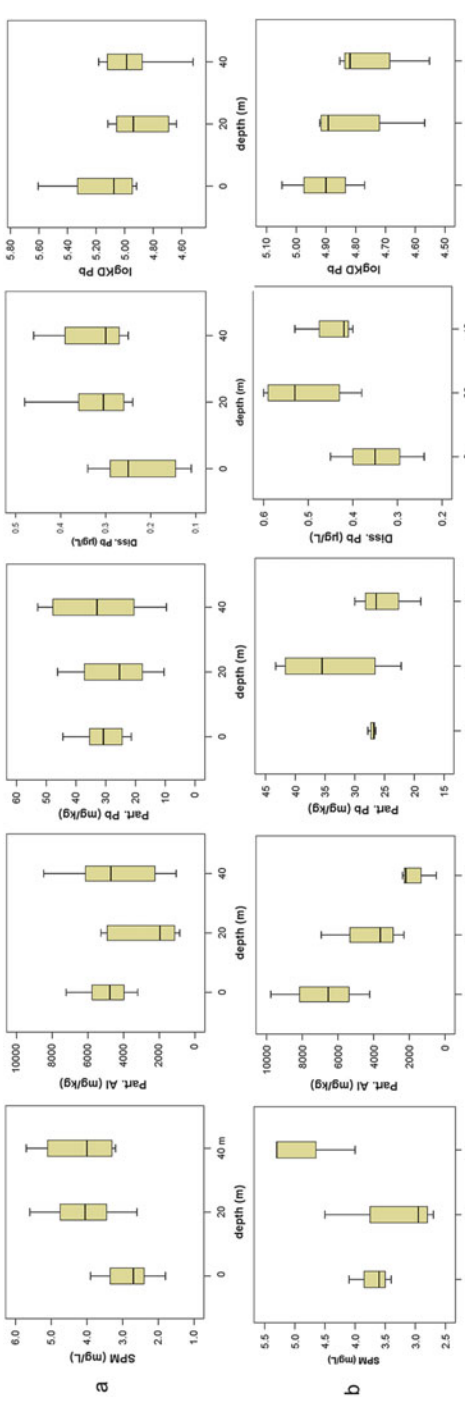


Fig. 11 Variations of SPM, particulate Al and Pb (w/w) dissolved Pb and K_D for Pb in the water column of outer marine stations of the Acheloos River during (**a**) a spring and (**b**) a summer month

colloidal and dissolved forms could also enhance desorption from suspended solids [3]. Desorption is a slow process and is greatly affected by the size of the estuary [13]. Therefore, any further changes of the hydrological regime of the Acheloos River may have a great impact on the behaviour and fate of trace metals, not only in the fresh–saline water interface, but in the coastal marine environment as well.

Surface and core sediments were collected from the upper and lower part of the Acheloos Estuary (Fig. 2). A detailed description of textural and mineralogical composition, as well as information on the sampling network, is given in Dassenakis et al. [25]. Table 5 summarises the results of extractions by conc. HNO_3 and 0.5 N HCl of the surface sediment samples of the Acheloos Estuary in the aforementioned zones. Compared to the composition of the continental crust (UCC; [61]; Table 3) much lower Al, Fe levels are detected in the Acheloos system, because of the incomplete dissolution of the crystal lattice of aluminosilicates by HNO_3. In general, all other metals are lower than the UCC values, reflecting low levels of pollution, due to the absence of major urban and industrial waste water discharges from point sources.

For a better consideration of the spatial variation, the estuary is subdivided into the following zones: zone A represents the upper estuary where the upper boundary of the salt wedge is observed during the low flow seasons; zone B extends over the area where the salt wedge is observed most of the time; zone C consists of a single station located at the river mouth and the sill bar; zone D is the lower estuary, off the river's mouth; zone E consists of the offshore station of the lower estuary at the boundaries with the marine sector, where salinity exceeds 25 and the water depth is $>$20 m.

Average contents of carbonates and organic carbon (OC) in the five zones of the system are 27% and 2.3% (zone A); 38% and 1.2% (zone B); 25% and 2.5% (zone C); 51% and 0.5% (zone D); 24% and 2.1% (zone E), respectively. Apparent differences on the composition of sediments exist between the zones. Sediments of zones B and D contain higher amounts of carbonates, and lower amounts of organic carbon and Al, whereas sediments of zones A, C and E contain lower amounts of carbonates and higher amounts of organic carbon and Al. Aluminium, Fe, Cu, Ni, Pb and Zn follow a very similar spatial distribution pattern (range of Spearman correlation coefficients between Al and other metals r: 0.714–0.928). On the contrary, Mn follows the distribution of carbonates ($r = 0.633$; $p = 0.015$), therefore indicating that calcite minerals might act as nucleation centres for manganese oxides, since there is a microzone of higher pH on the surfaces of carbonates [25]. The identical patterns of distributions of OC, Al, Fe and trace metals signify that clay minerals, originating from soil erosion, serve as a host phase for OC, and Al, Fe rich phase [20]. These fine clays are effective carriers for trace metals [47]. Evidently, compositional differences play a significant role in the distribution patterns of "total" trace metals and after normalising to Al, the large variations between the zones become "smoother" to some extent. However, when considering the fraction of metals extracted by dil. HCl (Table 5), it is clearly illustrated that zones A, C and E are accumulation sites of labile metals.

The increased labile metal contents in zone E compared to zone D indicate that a significant part of metals is flushed out of the estuary and settles in the offshore stations. Increased labile metal contents at zone A, where the end of the salt wedge is occasionally observed, could be ascribed to coagulation of particles in the "salt-stress" interface and subsequent sedimentation. Hydrodynamics and the geomorphology of the system also play a significant role. Zone C, which is a zone of trace metal accumulation, is located landward from the sill bar that could act as a natural barrier of particulates enriched by geochemical processes, at least under regular flow conditions. These particles could be recycled back to zones A and B at periods of low river discharge when seawater intrudes the upper part of the estuary, or flushed away under high river discharge conditions.

The impacts of changing hydrological regime on the distribution of metals, as well as of natural and/or anthropogenic activities in the Acheloos catchment area, could be depicted from the study of cored sediments.

Magnetic parameters in the bulk cored sediments of the Acheloos Estuary were studied by Scoullos and Oldfield [46] and allowed for the recognition of two major magnetic components: the first one of high magnetic concentrations and relatively softer demagnetisation behaviour corresponds to clay-like sediments, whereas the second population of particles, which corresponds to coarser, sandy sediments, appears to be richer in canted anti-ferrimagnetic grains (e.g. hematite) and probably has a deeper subsoil or bed-rock origin. Comparison of the vertical distributions of volume-specific magnetic susceptibility with 0.5 N HCl leachable fractions of Cu and Zn showed close similarities in their profiles in the clay-size sediments of (a) the lower reaches of the estuary upstream from the river mouth, and (b) in the upper layers (>12 cm depth) of the cores obtained from the mouth bar of the river. In the latter area, sediments of deeper layers are coarser, poorly correlated to the leachable fractions of metals and most probably reflect reworked littoral material from a different provenance, or an early denudational regime [46].

Figure 12 illustrates the vertical distributions of labile fractions of metals from a short core obtained from zone B, normalised to Al in order to compensate for grain-size effects. Sediments of the upper strata have higher metal contents or higher metal to Al ratio values. The same pattern is also observed in other cores obtained from the Acheloos Estuary [25]. Apart from direct, recent anthropogenic activities in the entire catchment area, another cause of the observed increase in the surface sediments seems to be the reduction of particulate load of the estuary. Before the construction of dams the inert, non-polluted matter, which originated from rock weathering and natural soil erosion, was mixed with authigenic precipitates, which originated from the reactions in the dissolved and particulate phase under changing physicochemical conditions. This authigenic suspended matter contains significantly higher concentrations of pollutants. After the construction of dams, there was a diminished frequency of floods capable of flushing out the system by transporting newly deposited authigenic sediments from the estuary to the sea. By changing the nature and the relative contribution of particles in the mixture of suspended sediments coupled with the minimisation of the fraction which comes from the mountains, the final composition of the naturally generated suspended matter becomes increasingly loaded with pollutants, such as trace metals.

The fractionation of metals in the sediments of zones A to E, following the sequential extraction procedure presented in Table 1, are shown in Fig. 13. More

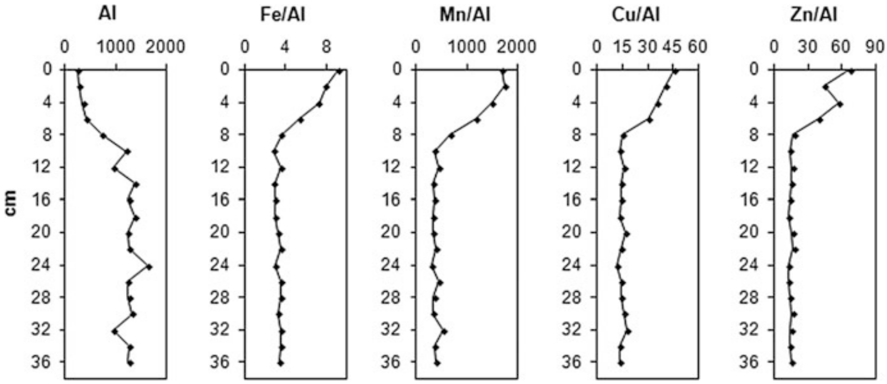


Fig. 12 Vertical distribution of 0.5 N HCl extractable Al and normalised metal ratios to Al, in cored sediments obtained from zone B

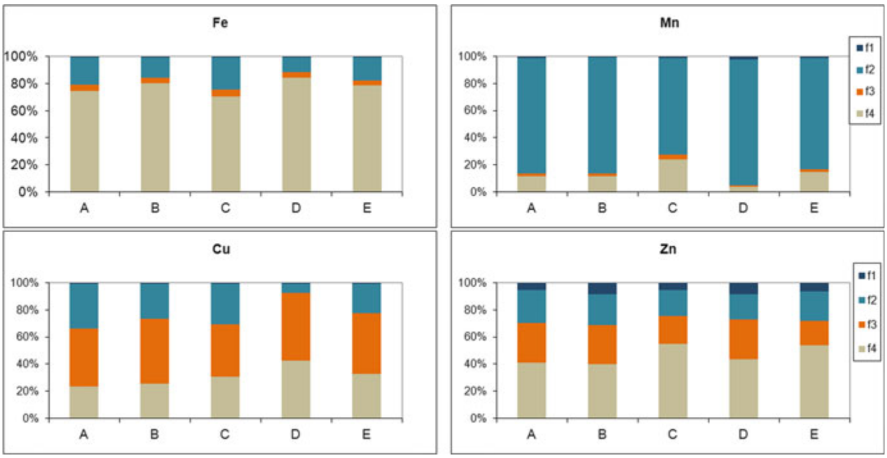


Fig. 13 Percentages of the geochemical fractions of metals in the surface sediments of zones A–E of the Acheloos system. The fractions of metals are F1: easily extractable; F2: non-lattice held, inorganic; F3: organic; F4: residual and are determined by following the sequential extraction procedure of Table 1

than 70% of "total" Fe content is lattice held. Manganese is mainly found in the non-lattice held inorganic fraction, which accounts for 71–93% of Mn total content, suggesting its presence as oxyhydroxides and bound to carbonates. The proportions of Cu and Zn are also elevated in the non-lattice held inorganic fraction consisting on average 23.6% and 21.4% of the "total" metal content. This indicates an increased mobility of these metals in the Acheloos Estuary. However, the dominant component of the "total" metal content is the non-lattice held organic fractions of Cu and Zn contributing on average 45.2% and 25.4% to the "total" contents.

A note should be made at this point about the limitations of sequential extraction procedures, including the non-selectivity of reagents, the potential redistribution of metals among phases during extraction, or incomplete extractions [62]. The non-selectivity of reagents to attack only one solid phase is more profound when dealing with anoxic sediments, when metal sulfides are likely to occur. For example, Shannon and White [63] conducted laboratory experiments with known additions of synthetic iron oxyhydroxides (FeOOH), iron monosulfide (FeS) and pyrite (FeS_2) to natural freshwater lake sediment. The authors found that the extraction by the 0.04M $NH_2OH \cdot HCl$ in 25% acetic acid reagent at 96°C removed 25% of the Fe added as FeS, but it did not dissolve FeS_2. Furthermore, the same reagent was found to extract 50% of Zn which was present as ZnS [64]. In the case of the Acheloos sediments, we have no indication of anoxia (smell, black colour bands). Mineralogical analysis by X-ray diffraction (XRD) analysis showed that the main minerals present in the sediments were calcite, quartz, biotite, as well as illite and chrorite [25]. Therefore, metal sulfide minerals, if present, would exist in very small amounts, mainly in the subsurface sediments. In this case, these phases are expected to be extracted in the F2 fraction (the non-lattice held inorganic fraction) of the sequential extraction procedure.

Characteristic distributions of the exchangeable, inorganic and organic and the residual fractions of Fe and Cu in surface and subsurface sediments of cores obtained from the five zones of the system are presented in Fig. 14.

Comparison of the surface and subsurface sediments of the two cores obtained from zone A shows substantial variations with depth of labile Fe contents, dominated by Fe oxyhydroxides, and probably trace amounts of Fe sulfides, formed in situ by redox reactions. These differences suggest that the amount of Fe oxyhydroxides leached from soils of the upper catchment area and transported and subsequently accumulated in the upper estuary has changed over time, due to the combination of land-use changes and river discharge. In situ reductive dissolution of Fe oxyhydroxides is expected to contribute only to a small extent, hence, the variations of the non-lattice held inorganic fraction primarily reflect the long-term modification of oxyhydroxides inputs in this zone.

Moving downstream to zone C, at the sill bar, surface sediments contain much higher amounts of Fe oxyhydroxides than the subsurface ones, reflecting the relative enrichment of sediments by the authigenically formed oxyhydroxides favoured by the prolonged residence time of waters due to the reduction of river discharge and the influence of the salt wedge. The relative depletion of Fe oxyhydroxides at the subsurface sediments of zone C and the relative enrichment at the subsurface sediments of zones D and E signify that in the past, under a different hydrological regime of higher river discharge, the labile fraction of Fe was flushed away, off the river's mouth and accumulated in the offshore stations of the lower estuary. The same processes explain the relative depletion of labile forms of Cu in the subsurface sediments in the cores of zones C and D. In the surface sediments of zone D, an increase of the organic fraction of both Fe and Cu is observed. Different transport pathways of colloidal and particulate metals during their passage through the active mixing zone [20] or an in situ control (biological uptake) of an autochthonous component of organic matter [4] could be the possible reasons. The overall much lower amounts of all fractions of metals in the surface

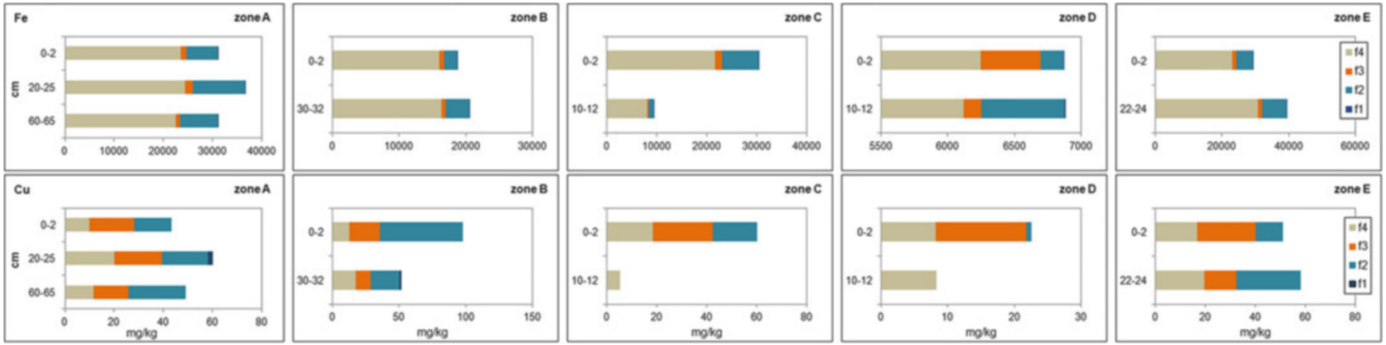


Fig. 14 Geochemical fractionation of Fe (*upper panel*) and Cu (*lower panel*), employing the sequential extraction procedure (1) described in Table 1, in cored sediment obtained from zones A (*left*) to zone E (*right*). Fraction F1: easily extractable; F2: non-lattice held, inorganic; F3: organic; F4: residual

and subsurface sediments of zone D in relation to other zones are attributed to the dilution due to the high carbonates contents, which in this zone are twice higher than in the other zones.

At the offshore station of zone E, the increased contents of labile Fe (and Cu to a lesser extent) of the subsurface sediments in relation the surface ones reflects more pronounced transport and deposition processes at this site in the past.

Putting together the variations of geochemical partitioning in the temporal and spatial scale, it can be proposed that there is propagation of the active mixing zone towards the inner part of estuary, where particles enriched in trace metals are deposited. This observation should be combined with the prediction ([34] and the references therein) that under reduced discharges of sediment, the sandy beaches and island barriers of the Acheloos delta will gradually erode and coastal lagoons will be intruded by seawater. These findings are important for the design of appropriate management measures of the area.

5 General Discussion and Conclusions

The Louros Estuary, which has a relatively narrow intermixing zone, low river flow and insignificant tides, has a saline water wedge intruding along the river and forming a small water mass, different from the riverine and the marine ones. The concentrations of dissolved and particulate metal species in this landward moving water body are considerably higher than in the other water masses. Physical entrapment of suspended solids landwards of the sill bar and the long residence time of suspended particles during periods of low flow regime, as well as re-suspension of sediments and subsequent benthic fluxes of metals or desorption from the suspended sediments, favour release processes of metals into solution. In the same region, new particles are formed authigenically, by precipitation of iron and manganese oxides and coagulation of clay minerals, at the interface of fresh and saline water. The accumulation of particulate metals in the salt wedge affects directly their distribution in the sediments, where the concentrations of some metals, including Al, exhibit their maxima. Although the zone of the salt wedge formation acts as trap of trace metals, the system should be considered as a periodic, (potential) source of metals for the adjacent sea, the Amvrakikos Gulf, particularly during flooding episodes.

Furthermore, the small size of the estuary, the shallowness and the low river flow indicate the vulnerability of its present, natural structures against changes. Pressures of local or global character could easily affect the borders of the fresh–saline water interface and transform the Louros Estuary into a secondary source of metals.

In the case of the much bigger and highly fragmented Acheloos River, the study reveals that the fresh–saline water interface is enriched in trace metals by re-suspension of sediments and desorption processes. Desorption of trace metals from the particles occurs in the intermixing zone, as well as in the offshore surface and subsurface marine waters, at 20 and 40 m depth, due to competing and complexing effects of major seawater ions. Currently, a significant fraction of labile metals is accumulated in the upper part of the estuary, the sill bar and the offshore stations of the lower estuary. Although the river is not heavily polluted, temporal trends depicted from the study of sediment cores show that surface sediments are enriched in labile forms of metals, as a result of non-point anthropogenic sources, in combination with a considerable decrease of the water flow and floods (due to the construction and operation of dams), which in the past used to flush out recently deposited polluted sediments, "diluting" the authigenic and anthropogenic component by inert, non-polluted particles.

Modifications of the hydrological regime have resulted in the transfer of the intermixing zone towards the inner part of the estuary, where authigenic, rich in metals, particles are deposited. At the same time, decreased river discharges and sediment fluxes have narrowed the zone at which transport of labile metals takes place under regular flow conditions.

A further decline in water and sediment fluxes, in case of the proposed diversion of a portion of Acheloos waters towards Pinios river is realised, or due to climatic changes, may result in the development of a much stronger (in length and duration) salt wedge, intensifying the processes that are responsible for the relative enrichment of the trace metals in the estuarine zone. Then, episodic flooding of the estuary could release the finest, unconsolidated particles enriched in trace metals into the adjacent wetlands and the sea with a potential impact also on the biota through the food chain.

The results of the research carried out in the two unique, fresh–saline water interface systems are important not only in order to inform us about the geochemical processes in nature, but also in order to provide the necessary knowledge to properly manage these systems for the benefit of the environment and the sustainable development of the impacted areas. The existing provisions of the EU Water Framework Directive and the EU Marine Strategy Framework Directive suggest due attention to the processes in transitional zones described in this work. Furthermore, the Barcelona Convention Protocol for the Integrated Coastal Zone Management, as well as the Ecosystem Approach, point to the need for a thorough and systematic integration of research results into a combined management of coastal zones and water resources.

References

1. Bewers JM, Yeats PA (1989) Transport of river-derived trace metals through the coastal zone. Neth J Sea Res 23(4):359–368
2. Ollivier P, Radakovitch O, Hamelin B (2011) Major and trace element partition and fluxes in the Rhône River. Chem Geol 285:15–31
3. Roussiez V, Ludwig W, Radakovitch O, et al. (2011) Fate of metals in coastal sediments of a Mediterranean flood-dominated system: an approach based on total and labile fractions. Estuarine Coastal Shelf Sci 92(3):486–495

4. Guieu C, Martin JM (2002) The level and fate of metals in the Danube River Plume. Estuarine Coastal Shelf Sci 54(3):501–512
5. Jiann KT, Wen LS, Santschi PH (2005) Trace metal (Cd, Cu, Ni and Pb) partitioning, affinities and removal in the Danshuei River estuary, a macro-tidal, temporally anoxic estuary in Taiwan. Mar Chem 96:293–313
6. Shiller AM, Boyle EA (1991) Trace elements in the Mississippi River delta outflow region: behavior at high discharge. Geochim Cosmopolitica Acta 55:3241–3251
7. Zhou JL, Liu YP, Abrahams PW (2003) Trace metal bahaviour in the Conwy Estuary, North Wales. Chemosphere 51:429–440
8. Turner A, Millward GE, Le Roux SM (2004) Significance of oxides and particulate organic matter in controlling trace metal partitioning in a contaminated estuary. Mar Chem 88:179–192
9. Zwolsman JJG, van Eck GTM (1999) Geochemistry of major elements and trace metals in suspended matter of the Scheldt Estuary, Southwest Netherlands. Mar Chem 66(1–2):91–111
10. Horowitz AJ, Elrick KA (1987) The relation of stream sediment surface area, grain size and composition to trace element chemistry. Appl Geochem 2:437–451
11. Radakovitch O, Roussiez V, Ollivier P, et al. (2008) Input of particulate heavy metals from rivers and associated sedimentary deposits on the Gulf of Lion continental shelf. Estuarine Coastal Shelf Sci 77:285–295
12. Roussiez V, Probst A, Probst JL (2013) Significance of floods in metal dynamics and export in a small agricultural catchment. J Hydrol 499:71–81
13. Morris AW (1990) Kinetic and equilibrium approaches to estuarine chemistry. Sci Total Environ 97(98):253–266
14. Elbaz-Poulichet F, Garnier JM, Guan DM, et al. (1996) The concervatiive behaviour of trace metals (Cd, Cu, Ni and Pb) and As in the surface plume of stratified estuaries: Example of the Rhone River (France). Estuarine Coastal Self Sci 42:289–310
15. Fu J, Tang X, Zhang J, Balzer W (2013) Estuarine modification of dissolved and particulate trace metals in major rivers of East-Hainan, China. Cont Shelf Res 57:59–72
16. Turner A, Millward GE (2002) Suspended particles: their role in estuarine biogeochemical cycles. Estuarine Coastal Shelf Sci 55:857–883
17. Puig P, Palanques A, Sanchez-Cabeza JA, Masque P (1999) Heavy metals in particulate matter and sediments in the southern Barcelona sedimentation system (North-western Mediterranean). Mar Chem 63:311–329
18. Zwolsman JJG, van Eck GTM, Van Der Weijden CH (1997) Geochemistry of dissolved trace metasl (cadmium, copper, zinc) in the Scheldt Estuary, Southwestern Netherlands: Impacts of seasonal variability. Geochim Cosmochim Acta 61(8):1635–1652
19. Nolting RF, Helder W, De Baar HJW Gerringa LJA (1999) Contrasting behaviour of trace metals in the Scheldt estuary in 1978 compared to recent years. J Sea Res 42:275–290
20. Wen LS, Warnken KW, Santschi PH (2008) The role of organic carbon, iron, and aluminium oxyhydroxides as trace metal carriers: comparison between the Trinity River and the Trinity River Estuary (Galveston Bay, Texas). Mar Chem 112:20–37
21. Bibby RL, Webster-Brown JG (2005) Characterisation of urban catchment suspended particulate matter (Auckland region, New Zealand); a comparison with non-urban SPM. Sci Total Environ 343(1–3):177–197
22. Warren LA, Haack EA (2001) Biogeochemical controls on metal behavior in freshwater environments. Earth-Sci Rev 54(4):261–320
23. CIESM (2006) Fluxes of small and medium-size Mediterranean rivers: impact on coastal areas. Ciesm Workshop Monographs No. 30, Monaco
24. Haralambidou K, Sylaios G, Tsihrintzis VA (2010) Salt-wedge propagation in a Mediterranean micro-tidal river mouth. Estuarine Coastal Shelf Sci 90:174–184
25. Dassenakis M, Degaita A, Scoullos M (1995) Trace metals in sediments of a Mediterranean estuary affected by human activities (Acheloos river estuary, Greece). Sci Total Environ 168:19–31

26. Scoullos MJ, Pavlidou AS (2003) Determination of the lability characteristics of lead, cadmium and zinc in a polluted Mediterranean brackish-marine interface system. Water Air Soil Pollut 147(1–4):203–227
27. Scoullos MJ, Pavlidou AS (2000) Metal speciation studies in a brackish/marine interface system. Global Nest: Int J 2(3):255–264
28. Scoullos M, Dassenakis M, Zeri C (1996) Trace metal behavior during summer in a stratified Mediterranean system: The Louros estuary (Greece). Water Air Soil Pollut 88(3–4):269–295
29. Scoullos M, Zeri C (1993) The use of mineral magnetic measurements to study the transport and sedimentation of particles in the Gulf of Lions (NW Mediterranean). Oceanol Acta 16 (1):53–61
30. Scoullos M, Dassenakis M, Zeri C (1990) Study of the transport of metals in the estuary of Louros River during summer. In: Third hellenic symposium on oceanography and fisheries, Abst. Hellenic Centre for Marine Research (HCMR), Athens, pp 366–374
31. Scoullos M, Dassenakis M, Zeri C et al (1987) Chemical studies of main estuaries and coastal areas of Greece. Progress Report to DGXI, Project GEC-ENV-560 GR
32. Dassenakis M, Scoullos M, Gaitis A (1997) Trace metals transport and behaviour in the Mediterranean estuary of Acheloos River. Mar Pollut Bull 34(2):103–111
33. Kapsimalis V, Pavlakis P, Poulos SE, et al. (2005) Internal structure and evolution of the late quaternary sequence in a shallow embayment: The Amvrakikos Gulf, NW Greece. Mar Geol 222–223:399–418
34. Skoulikidis N (2009) The environmental state of rivers in the Balkans—A review within the DPSIR framework. Sci Total Environ 407:2501–2516
35. Kousouris T, Diapoulis A, Bertahas E, Gritzalis K (1989) Proceedings of the conference on environmental science and technology. Aegean University, Mytilini, pp. 300–309
36. Scoullos M, Dassenakis M, Gaitis A (1992) Distribution of nutrients in the Acheloos river mouth, Greece. Rapp Comm Int Mer Med 33:182
37. Skoulikidis N (2003) Hydrochemical character and spatiotemporal variations in a heavily modified river of western Greece. Environ Geol 43:814–824
38. Scoullos M, Maroulakou-Christodoulou M, Dassenakis M (1985) Preliminary results on the nutrient distributions in the Patraikos Gulf and the Acheloos Estuary. Rapp Comm Int Mer Medit 297:51–52
39. Anonymous (1992) The Greek river dilemma. Mar Pollut Bull 24(5):225
40. Close DH (1998) Environmental NGOs in Greece: the Acheloos campaign as a case study of their influence. Environ Polit 7(2):55–77
41. Scoullos M (1996) Environmental impacts of the Acheloos river diversion. Proceeding of the 15th congress of the international association of bridge and structural engineering, Copenhagen, pp 61–77
42. Mackereth FGH (1969) A short core sampler for sub aqueous deposits. Limnol Oceanogr 14:145
43. Riley JP, Taylor D (1968) Chelating resins for the concentration of trace elements from sea water and their analytical use in conjunction with atomic absorption spectrometry. Anal Chim Acta 40:479–485
44. Scoullos M, Dassenakis M (1984) Proceedings of the first hellenic symposium on oceanography and fisheries pp 302–309 (in Greek)
45. Agemian H, Chau ASY (1976) Evaluation of extraction techniques for the determination of metals in aquatic sediments. Analyst 101:761–767
46. Scoullos MJ, Oldfield F (1986) Trace metal and magnetic studies of sediments in Greek estuaries and enclosed gulfs. Mar Chem 18:249–268
47. Salomons W, Förstner U (1984) Metals in the hydrocycle. Springer–Verlag, Berlin
48. Kersten M, Smedes F (2002) Normalization procedures for sediment contaminants in spatial and temporal trend monitoring. J Environ Monit 4:109–115
49. Gaudette HE, Flight WR, Toner L, Folger DW (1974) An inexpensive method for the determination of organic carbon in recent sediments. J Sediment Petrol 44:249–253

50. Scoullos M, Botsou F, Zeri C (2014) Linking environmental magnetism to geochemical studies and management of trace metals. Examples from fluvial, estuarine and marine systems. Minerals 4:716–745
51. Viers J, Dupré B, Gaillardet J (2009) Chemical composition of suspended sediments in World Rivers: new insights from a new database. Sci Total Environ 407:853–628
52. Balls PW (1989) The partition of trace metals between dissolved and particulate phases in European coastal waters: a compilation of field data and comparison with laboratory studies. Neth J Sea Res 23(1):7–14
53. Rondeau B, Cossa D, Gagnon P, et al. (2005) Hydrological and biogeochemical dynamics of the minor and trace elements in the St. Lawrence River. Appl Geochem 20:1391–1408
54. Sholkovitz ER, Boyle EA, Price NB (1978) The removal of dissolved humic acids and iron during estuarine mixing. Earth Planet Sci Lett 40:130–136
55. Laslett RE, Balls PW (1995) The behaviour of disolved Mn, Ni, and Zn in the forth, an industrialized, partly mixed estuary. Mar Chem 48:311–328
56. Santos-Echeandía J, Prego R, Cobelo-García A, Millward GE (2009) Porewater geochemistry in a Galician Ria (NW Iberian Peninsula): Implications for benthic fluxes of dissolved trace elements (Co, Cu, Ni, Pb, V, Zn). Mar Chem 117(1–4):77–87
57. Benoit G, Oktay-Marshall SD, Cantu A, Hood EM, Coleman CH, Corapsioglu MO, Santschi PH (1994) Partitioning of Cu, Pb, Ag, Zn, Fe, Al and Mn between filter-retained particles, colloids, and solution in six Texas estuaries. Mar Chem 45:307–336
58. Turner A, Millward GE, Bale AJ, Morris AW (1993) Application of the KD concept to the study of trace metal removal and desorption during estuarine mixing. Estuarine Coastal Shelf Sci 36:1–13
59. Benoit G, Rozan TF (1999) The influence of size distribution on the particle concentration effect and trace metal partitioning in rivers. Geochim Cosmopolitica Acta 63:113–127
60. Sholkovitz ER (1978) The flocculation of dissolved Fe, Mn, Al, Cu, Ni, Co and Cd during estuarine mixing. Earth Planet Sci Lett 41(1):77–86
61. Rudnick RL, Gao S (2003) Composition of the continental crust In: Rudnick RL (ed) Treatise on geochemistry, vol 3. Elsevier, Oxford, pp 1–64
62. Bacon JR, Davidson CM (2008) Is there a future for sequential chemical extraction? Analyst 133:25–46
63. Shannon RD, White JR (1991) The selectivity of a sequential extraction procedure for the determination of iron oxyhydroxides and iron sulfides in lake sediments. Biogeochemistry 14:193–208
64. Peltier E, Dahl AL, Gaillard JF (2005) Metal speciation in anoxic sediments: when sulfides can be construed as oxides. Environ Sci Technol 39:311–316

The Evrotas River Basin: 10 Years of Ecological Monitoring

Ioannis Karaouzas, Christos Theodoropoulos, Leonidas Vardakas, Stamatis Zogaris, and Nikolaos Skoulikidis

Abstract This chapter is the outcome of a 10-year ecological monitoring survey in the Evrotas River Basin (ERB). Synthesising the main outcomes of past and ongoing research projects, it presents an overview of the basin's geographical, geological, hydrological and ecological features, focused on the ecological status according to the Water Framework Directive 2000/60/EC, and assesses the degree of environmental degradation caused by the major pollution sources and other anthropogenic pressures. Chemical, hydromorphological and biological data from studies carried out in the ERB during the past decade are integrated to derive spatial and temporal trends in environmental degradation. Despite the numerous sources of organic and inorganic pollution, which include, inter alia, olive mill and fruit juice processing wastewaters and agricultural, industrial and urban runoffs, the overall ecological degradation of the ERB is assessed as moderate and is located mainly at the downstream half of the basin, where the anthropogenic activities become intensified. However, the major impact in the ERB during the last decades has been the over-exploitation of the surface and groundwater resources for irrigation, which has resulted in the artificial desiccation of large parts of the basin's hydrological network. Despite the aforementioned issues, the aquatic benthic biota of the basin shows high resilience, but the fish fauna is severely affected by hydrological and morphological alteration. Biomonitoring, conservation and management responses to drought and pollution require approaches, which account for spatial and temporal variability. Within this perspective, a programme of measures is proposed, aiming at preserving and restoring the basin's water resources and aquatic ecosystem.

I. Karaouzas (✉), C. Theodoropoulos, L. Vardakas, S. Zogaris, and N. Skoulikidis
Hellenic Centre for Marine Research, Institute of Marine Biological Resources and Inland Waters, 46.7 km Athens-Sounio Ave., Anavyssos 19013, Greece
e-mail: ikarz@hcmr.gr

N. Skoulikidis et al. (eds.), *The Rivers of Greece: Evolution, Current Status and Perspectives*, Hdb Env Chem (2018) 59: 279–326, DOI 10.1007/698_2017_472,

Keywords Biota, Climate change, Drought, Ecological status, Intermittent, Mediterranean

Contents

1 Introduction and History

1.1 The History and Myths Associated with the River

Dating back to ancient times, according to the Greek geographer Pausanias (c. 110–180 A.D.), the first inhabitant of the Evrotas valley was Lelex, the king of the Leleges, an autochthonous pre-Hellenic tribe from the eastern Aegean area. Evrotas was formed into a river by King Lelex's grandson, the King Evrotas of Laconia, 'who channelled away the marsh-water from the plains by cutting through to the sea and when the land was drained, he called the created river Evrotas' [103]. After the death of King Evrotas, Lacedaemon who was married with the king's only daughter, Sparta, inherited the kingdom and raised a city near the banks of river Evrotas,

Fig. 1 The members of the French Scientific Expedition of the Peloponnese are passing over the former Kopanos bridge north of Sparta. The bridge survived until the first decade of the twentieth century. Pont de Eurotas by Prosper Baccuet (Expédition scientifique de Morée – Atlas, 1835, pl. XXVIII)

naming it Sparta. Since then, the river has been a long-standing historical symbol and a valuable economic resource for the city and the surrounding settlements, providing food, water and other goods and services to its people.

1.2 Environmental Research and Conservation History

The history of environmental research in the Evrotas River is fairly recent yet intense. The first collections of modern natural history data and a brief description of the river were made in 1829 and are documented in the multivolume work presenting the results of the Expédition scientifique de Morée (Fig. 1) [1]. This expedition studied various aspects of the natural history of the Peloponnese for the first time in the modern period. However, since the Middle Ages, the Evrotas valley has been a world-renowned area for archaeology, focusing on Sparta and the surrounding region of Laconia. The bibliography, including many engravings and travellers' accounts, on archaic and classical Sparta and Laconia is remarkably rich.

There is in fact such a proliferation of archaeological and historical research in this area that Barnes [2] has even used the term ‘Spartanology’ to refer to the richness and breadth of research in the area. Laconophilia refers to the admiration of Sparta and the ancient Spartan culture, and this has attracted research interest since the classical times and seems to be continuing. Some of this work has included brief researches on landscape, geography and recent history as well [3, 4]; however, it seems that the area’s importance in antiquity has overshadowed its outstanding ecological interest. Environmental research is surprisingly recent, despite the extended areas with acknowledged biodiversity richness.

Environmental and nature conservation interests were initially focused on the Evrotas surrounding mountains, Taygetos and Parnonas. The distinctive lowlands and mid-elevation areas of the valley and its delta with their rich cultural landscapes [5] were not a focus of early research; the towering mountain peaks and gorges became well known for their high floral richness and high degree of plant endemicity [6, 7]. The importance of the area’s largest wetland, the Evrotas Delta attracted interest for its birdlife and the protected area encompassing the deltaic plain and the coast of Evrotas, was first delineated in 1995 as a proposed site of community interest within the Natura 2000 network of protected areas [8]. Despite the interest in the Evrotas’ riverscapes, limited work has been carried out for restoration or improvement [9]. However, extended LIFE-Nature projects (1997–2001) entitled ‘Implementation of Management plans for Pylos Lagoon and Evrotas Delta, Natura 2000 Sites, Greece’, promoted by conservation groups, such as the Hellenic Ornithological Society, as well as the LIFE-Environment project, promoted primarily by research organisations, are important milestones in the ecological research around the Evrotas River Basin (ERB).

Surprisingly, the interest in delineating and promoting protected areas within the river valley itself has been low, despite the outstanding riparian woodland and other habitats [10]; boundary extensions of the area’s Natura 2000 network were proposed and scientifically promoted solely due to the river’s ichthyological interest in 2015 [11]. It has been widely accepted that the protected area delineations in Greece may require further revision, and this is certainly the case in the Peloponnese [12]. In the complex multifunctional landscapes of the Peloponnese, selection of protected areas and delineation of boundaries should be evaluated by using a variety of taxonomic groups and/or biodiversity surrogates (e.g. vegetation and landscape types) as well as other criteria (cultural landscape quality, socio-economic criteria, etc.). The establishment of micro-reserves and specific protected area extensions to encompass tributaries, springs and riparian zones may be a cost-effective measure for the conservation of endemic species or specific habitat types, especially where endangered populations or taxa assemblages occur outside the currently designated Natura 2000 sites.

During the past decade, several European-funded research projects have been carried out in the river basin, covering many aspects of its ecology, biogeochemistry, hydrology and conservation. The first project aiming to assess its ecological status by implementing the Water Framework Directive 2000/60/EC and providing conservation and restoration measures was the 4-year (2005–2009), LIFE-

Environment project entitled ‘Environmental Friendly Technologies for Rural Development’ [13]. The objective of this project was to develop and demonstrate a ‘toolbox’ of environmental friendly technologies for the minimization of point and non-point source pollution from agricultural activities and the integration of their design in the river basin management plan of the ERB and its coastal zone. The next project was the ‘wildfires’ project funded by the UK Natural Environment Research Council (NERC) ‘rapid response’ Grant (NE/F01273X/1). The project was carried out at the incinerated area of the Parnonas Mt. and aimed to (a) quantify the potential range of postfire hillslope sediment and phosphorus yields and (b) compare the bioavailability of phosphorus in burned and unburned sediment source material after the wildfires that took place in the ERB at the end of August 2007. Later on, from 2009 to 2011, the MIRAGE (Mediterranean Intermittent River ManAGEment) research project followed, aiming to provide specific key knowledge for a better assessment of the ecological integrity in Mediterranean temporary streams due to climate change and land use change.

Currently, the GLOBAQUA research project (managing the effects of multiple stressors on aquatic ecosystems under water scarcity) is taking place in the ERB, aiming at identifying the prevalence of, and interaction between, stressors under water scarcity in order to improve knowledge of relationships between multiple stressors and to assess how these interactions determine changes in the chemical and ecological status of water bodies, as well as to improve water management practices and policies. The project started in February 2014 and continues until January 2019.

2 Description of the River Basin

2.1 Geography and Climate

The ERB is a mid-altitude Mediterranean Basin located in the southeastern Peloponnese, at the southern part of geographical zone No. 3 [14], including parts of the Laconia and Arcadia Prefectures and covering an area of 2,418 km^2 (Fig. 2). The river originates from the mountain ranges of Taygetos (2,407 m.a.s.l.) and Parnonas (1,940 m.a.s.l.), its main stem crosses 90 km of semi-mountainous landscape and wide valleys, and it drains into the Laconian Gulf. The slopes at the larger part of the basin area (65%) are higher than 15%. A 24% of the area’s slopes ranges between 5 and 15%, and only an 11% has slopes less than 5%, indicating the rugged nature of the terrain. Elevations higher than 600 m.a.s.l. are observed at 41% of the basin area, a 46.2% of the area presents elevations ranging between 150 and 600 m.a.s.l., and only 12.7% has an elevation up to 150 m.a.s.l.

The ERB has a typical Mediterranean climate with mild and cool winters, followed by prolonged hot and dry summers. Seven meteorological stations are situated in the basin, recording an average annual temperature of 16°C, a mean

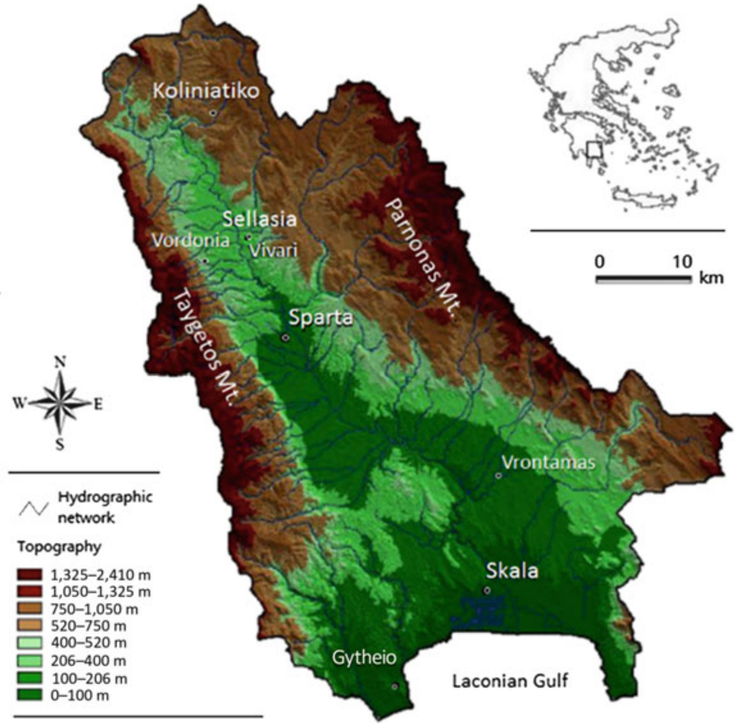


Fig. 2 The topography of the Evrotas River Basin

annual precipitation of 803 mm (2000–2008) ranging from 539 to 1,324 mm and a mean annual potential evapotranspiration of 668 mm [13]. Mean monthly temperatures typically range between 4 and 11°C during winter and between 22 and 29°C during summer [15]. The majority of rainfall occurs from October to March (70%), with the highest amount being recorded in November/December and the lowest in June. The recorded precipitation indicates a reduction from west to east and from north to south; the mid- and lower part of the Evrotas valley and the Laconian coast lie in a rain shadow area formed by the surrounding mountain massifs [16]. The highest precipitation is recorded in the Taygetos Mt. (1,300–1,600 mm), followed by the one in Parnonas (1,000–1,200 mm). An amount of 700–800 mm has been recorded in the lower parts of the basin and 500–550 mm in the coastal area. Precipitation shows varying interannual height; however, a decreasing trend has been observed during the last decades.

2.2 Geology

The mountainous area of the basin is formed by Mesozoic-Palaeogene limestones (42%) and impermeable rocks, such as flysch and schists (29%), while the lower parts of the valley are filled with Pliocene and Quaternary sediments forming extensive alluvial aquifers (Fig. 3). Almost all geotectonic units of the Peloponnese are included in the ERB, (a) the Plattenkalk or Mani Unit, (b) the Phyllite–Quartzite or Arna Unit, (c) the Tripolis Unit and (d) the Pindos Unit.

The Plattenkalk Unit represents the 'autochthonous' basement of the ERB. The stratigraphic bedrock of the unit, known also as phyllitic basement rocks (PBRs), consists of phyllites, schists, quartzites and meta-conglomerates. Over this unit lies a carbonate column, which consists of dolomites and crystalline limestone. The column continues with a formation comprised of siliceous schist, over which is the formation of Vigla. The column ends with crystalline platy limestone and multicoloured marbles, which lie near to the transition of the slightly metamorphosed flysch. The Plattenkalk Unit is of fairly low transformation degree. The current structure is complex and presents vertical transitions between PBRs and carbonate rocks. The Phyllite–Quartzite Unit (Arna), which overlies the Plattenkalk Unit, consists of metamorphic rocks, characterised by strong deformation of at least three phases. The rocks of the unit consist of schist, micaceous schist and quartzite, meta-conglomerates, mafic and ultramafic rocks. The Tripoli Unit lies over the Phyllite–Quartzite Unit. The lower part of this unit consists of slightly metamorphic formations, known as Tyros Beds, which lie under a sequence of carbonate rocks. This sequence of bituminous thick-bedded to unbedded limestones and dolomites is known as Tripolitza kalk. The upper part is the flysch of the unit, the sedimentation

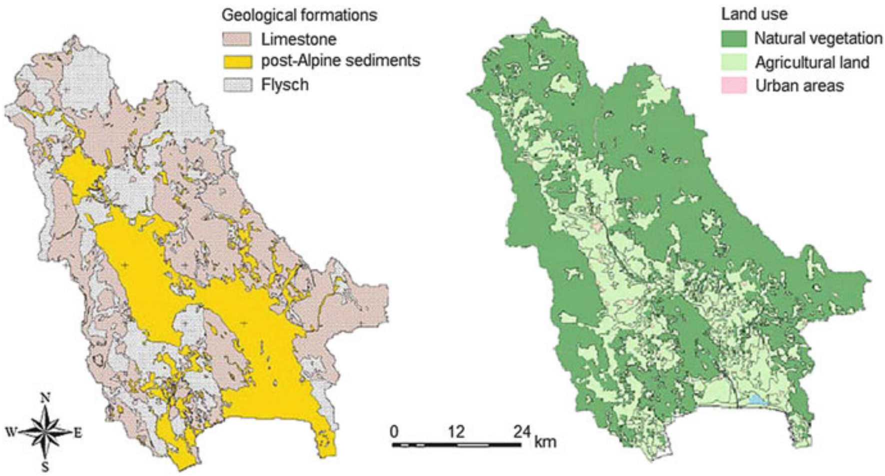


Fig. 3 Geological formations (*left*) and land cover/use (*right*) in the Evrotas River Basin

of which started in the Upper Eocene. In the ERB all the formations of Tripoli Unit are found. The Pindos Unit overlies the previous units. Only the upper section of Pindos Unit, known as Arcadian Nappe, is found in the ERB and more particularly the following formations: the lowest stratigraphic layer is a clastic sequence of sandstones, pelites and radiolarites in alterations with brecciated limestone of Cenomanian age, known as First Flysch. The overlaying formation consists of platy limestone with Globotruncanidae. Over the limestone lies flysch transitional beds and, finally, the flysch of the unit. Only the First Flysch and the upper cetaceous limestone appear in the area of interest.

2.3 Land Cover and Land Use

Most of the river basin's landscape is covered by seminatural areas comprising 61% of the total basin area, followed by agricultural areas (38%) and urban land (1%). The delineated CORINE land cover classes (CLC 2000, http://www.eea.europa.eu/publications/COR0-landcover) are scrub and/or herbaceous vegetation associations (60.8%), forests (16.0%), heterogeneous agricultural areas (15.0%), permanent crops (6.5%), open spaces with little or no vegetation (1.1%), urban fabric (0.3%) and arable land (0.1%), while the remaining 0.2% is industrial, commercial and transport units, mine, dump and construction sites and artificial, nonagricultural vegetated areas (Fig. 3).

Until the early twentieth century, most of the agriculture in the Evrotas valley was not irrigated, and the basin maintained a subsistent, low-intensity rural economy without extracting substantial surface and subsurface water resources [3, 17]. Today, the major anthropogenic pressures in the ERB are mainly derived from agriculture and livestock activities and include over-exploitation of water for irrigation, disposal of agro-industrial wastes and localised agrochemical pollution. However, the water mass balance in the ERB remains positive, and the requirements for irrigation and drinking water are met. Water stress problems are encountered during summer, due to the increasing irrigation demands, which may result to widespread desiccation of the Evrotas River network. There are many, mostly illegal, surface water abstraction points (permanent and temporary weirs) along the course of Evrotas and numerous private and municipal borehole drillings (estimated at around 3,500) for irrigational use, scattered over the entire basin. Irrigation withdrawals were estimated to 62 mm^3 from groundwater wells and 15 mm^3 from direct abstractions from the stream [18]. Regarding wastewater treatment, only one plant exists in the ERB, in the city of Sparta, while the villages are served by private permeable and impermeable cesspools.

2.4 Hydrological and Hydrogeological Features

The ERB is characterised as semiarid due to the low ratio (0.46) between the mean annual precipitation and the potential evapotranspiration (for semiarid zones: $0.20 \leq P/PET < 0.5$; [19]) during a hydrological period between 1998 and 2000. The Evrotas River is one of the last large free-flowing rivers in Greece; there are no dams, neither along its main course nor on its numerous tributaries. However, in the summer, when water flow becomes minimum, farmers build earthen weirs to use water for irrigation or to prevent seawater intrusion in the mouth of the river, so that several reaches become temporarily fragmented. According to historical information [20], the Evrotas River was once permanently flowing throughout the year, but nowadays it presents temporary flow characteristics [21]. Overall, the ERB is a complex hydrological system consisting of perennial, intermittent, ephemeral and episodic tributaries. Specific reaches of the Evrotas main stem dry out every summer, whereas others desiccate during particularly dry years. At the upstream portion of the catchment, from the Koliniatiko tributary to the upstream Vivari springs, the river dries out regularly for about 15 km, as a result of both downwelling processes in the alluvial aquifers and water abstractions for irrigation. In the downstream part of the catchment, the river passes through the Vrontamas Gorge, structured by karstified carbonates. There, during summer, the river becomes dry due to the infiltration of water in the karstic system of the Vrontamas Gorge, while further downstream, at the area of Skala, the river reappears as a result of significant karstic inputs from the adjacent Vasilopotamos springs.

The main tributaries of Evrotas are Oinous (drainage area, 349.8 km^2), Magoulitsa (46.5 km^2), Gerakaris (43.2 km^2), Kakaris (24.5 km^2), Rasina (55.8 km^2), Mariorema and Xerias (17.5 km^2, episodic flow). Most of these headwater streams of the ERB have an intermittent flow regime. The river is fed by numerous springs, which are the main contributors to its baseflow during the summer period; among them are a number of karstic springs at the upper- and mid-parts of the basin with significant inputs. Groundwater in the ERB is concentrated in karstic and alluvial aquifers, which communicate hydraulically. The alluvial aquifers are recharged by the river and by karstic inflows, which constitute an important water resource for the watershed. The karst aquifers occupy approx. 570 km^2 in the Taygetos and Parnonas mountains. In the lowlands, two main alluvial aquifer systems are located at the upstream (220 km^2) and the downstream parts of the river (275 km^2), respectively. Karstic inputs mainly originate from the aquifers of Taygetos Mt. The aquifers of the Parnonas Mt. do not contribute significant karstic discharges since the impermeable basis of the particular aquifers is encountered at high depths. The northern part of Parnonas feeds the upper Evrotas, while karstic waters from the central part of the mountain either discharge in the Vasilopotamos springs (near the town of Skala) or are lost in the sea [22]. The lowland alluvial aquifers are fed with water by the river.

The mean monthly discharge of the Evrotas main stem exhibits a gradual increase from summer towards early spring (March) and reaches a minimum in October (Fig. 4). Based on this variation, the river may be classified as a spring type

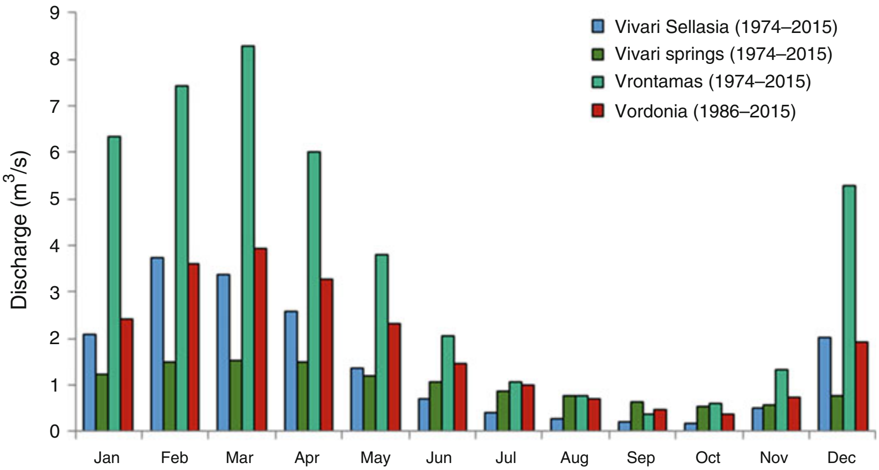


Fig. 4 Mean monthly discharge variation of Evrotas at upstream (Vivari), the midstream (Vordonia) and downstream (Vrontamas) reaches (data: Prefecture of Laconia)

(after the classification of Malikopoulos, in [23]); autumn and winter rainfalls are initially held within the extensive karstic aquifers of the basin, which after reaching their maximum capacity in March (when snow melts) they supply the river with maximum amounts of water. Minimum discharge is observed in October although autumn rainfalls start to express, since karstic and alluvial aquifers need first to be filled up. During summer/autumn months, the Evrotas main course dries out even at reaches where discharge gauging stations exist; at Vrontamas, prior to entering the homonymous gorge, the river has started to desiccate since 1990, when a severe drought wave affected Europe (end of the 1980s to beginning of the 1990s), and has been becoming dry in subsequent years. During the aforementioned drought period, the river desiccated even at reaches fed by substantial karstic inputs (e.g. Vivari).

Despite the high karstic inputs, which generally smooth discharge variations, the Evrotas River is prone to floods. The long-term (1974–2015) ratio between the monthly maximum and minimum discharge at Vrontamas is 21.8, which is one of the highest among the major Balkan rivers [24]. This ratio is, however, affected by two factors; maximum flows may have not been monitored due to technical restrictions, and minimum flows are triggered by water abstractions. Along the Evrotas main course, floods cause erosion of riverbanks and riparian areas and inundation of the lowlands around the river outflow. The areas at the feet of Taygetos Mt. around Sparta are also prone to floods. These floods are triggered by high slopes and increased sediment transport of the Taygetos Mt. tributaries [25]. The most recent catastrophic floods occurred in 1993–1994, 1999, 2000, 2005–2006 and 2016.

3 Ecological Features of the Evrotas River Basin

The ERB is considered a biodiversity hotspot in Greece [26–28], hosting high species richness in several taxonomic groups and habitat types, including many range-restricted and locally endemic plants, invertebrates and vertebrates and globally threatened species [8]. Parts of the wider Evrotas basin, particularly the Taygetos and Parnonas ranges, their rocky gorges and the dry southern parts of the Evrotas valley are known as one of two main endemism hotspot areas for an outstanding proportion of range-restricted plant species in the Peloponnese [12]. The high proportion of plant endemism is attributed to the relative isolation of the southern Peloponnese from the Balkan mainland, its southern position as a biogeographical refuge for many, so-called tertiary species that survived the Quaternary Ice Age, and a uniquely rich topography and habitat diversification [7]. There is high endemism of fish species in the southern Peloponnese, and although the vertebrate species richness is poor, their assemblage is unique in the Ionian freshwater ecoregion [29, 30].

3.1 Riparian and Aquatic Flora and Water-Dependent Vegetation

Historical records indicate that rich riparian forests were once thriving throughout the Evrotas watercourse. These forests, however, are nowadays restricted to specific patches concentrated mainly at the upper parts of the basin, but, still, the ERB hosts the most extensive and best-conserved lowland riparian forests of the Peloponnese. Many upland landscapes are still genuinely intact cultural landscapes with richly varied vegetation patterns [5]. The upstream riparian patches are dominated by oriental planes (*Platanus orientalis*), while at the mid-course, planes are often mixed with white willows (*Salix alba*), and within the Vrontamas Gorge, a variety of woody trees and shrubs create interesting riparian assemblages [10]. Small intermittent tributaries in the lowlands are mostly shrub dominated by *Nerium oleander* and *Vitex agnus-castus*; near the coast, *Tamarix* spp. are abundant, and extensive thickets of the giant reed cane *Arundo donax* are widespread. Other riparian vegetation patches in perennially flowing upland sections include alder woods (*Alnus glutinosa*), a locally scarce woodland community.

Due to the relatively smooth slopes of the main stem of the ERB, the river additionally hosts an interesting portion of hygrophilous taxa (*Potamogeton* spp.) and helophytes (*Nasturtium officinale*, *Lycopus europaeus*, *Mentha aquatica*, *Typha domingensis*, *Phragmites australis*) [31].

The Evrotas Delta, one of the Peloponnese peninsula's most important and extensive wetland habitats, was in former times a huge wetland basin, with extensive lagoons, multiple river tributaries and an array of permanent and temporary wetland habitat types, including a unique beach dune system [8]. During the 1950s, reclamation works began to drastically change the landscape. This resulted in the straightening and embanking of the Evrotas main stem and its spring-fed tributary, the Vasilopotamos. Nearly all the surrounding land was replaced by agriculture. For some time, rice fields, abandoned today, prevailed. Currently, olive groves and citrus orchards dominate the croplands, with several clusters of greenhouses. Only the thin coastal saline zone has been preserved to some degree; the abandoned rice fields, the extensive Asteri marshland and the Vivari lagoon sustain an important wetland hotspot, which is one of the largest remnant patches in the Peloponnese [32].

3.2 Stream Invertebrate Fauna

Taxa lists for many aquatic invertebrates have been poorly compiled for the ERB; knowledge at the species level is limited, even for taxonomic groups with high endemicity, such as freshwater gastropods [33]. The ERB shows some interesting faunal distinctions, including the presence of a unique gastropod species *Melanopsis praemorsa*, which in the Greek mainland has been recorded only in the Evrotas River [34]. A total of 96 benthic macroinvertebrate families have been found along the river basin, reflecting the diverse aquatic invertebrate fauna [35, 36].

Higher species richness occurs in the upper parts of the basin, with high abundance of Plecoptera, dominance of specific Ephemeroptera and Diptera species in the mid-reaches and, again, increased diversity (but not equal to the upper reaches and with different dominating classes) in the most downstream part of the ERB (Evrotas River delta). It must be noted that high abundance of Plecoptera and Trichoptera has been recorded mainly at the uppermost parts of the basin (Fig. 5).

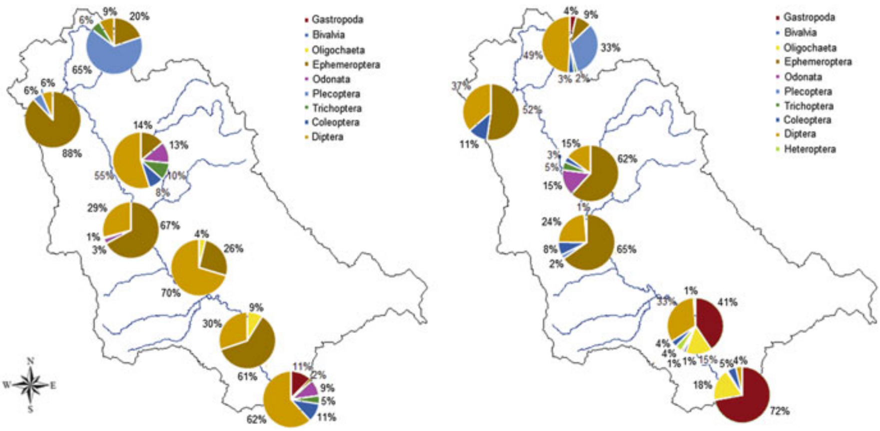


Fig. 5 Benthic macroinvertebrate distribution along the Evrotas basin during spring (*left*) and summer (*right*). Only major orders (>1%) are illustrated

During summer, diversity is reduced, and the abundance of Gastropoda, Heteroptera and Oligochaeta is increased at the most downstream parts of the river. However, a similar distribution pattern is observed, with Plecoptera thriving at the upper parts, being replaced by Ephemeroptera and Diptera at the mid-reaches.

Overall, throughout the basin, the most dominant benthic macroinvertebrate families (in terms of total species richness) are the Baetidae and Chironomidae, followed by Gammaridae and Ephemerellidae. Reaches with good ecological status, which are mainly located at the upper parts of the ERB, are mainly represented by species of the Baetidae, Chironomidae, Simuliidae, Ephemerellidae, Heptageniidae, Taeniopterygidae, Leuctridae, Perlodidae and Hydropsychidae families. Species of the Chironomidae, Baetidae, Simuliidae and Ephemerellidae families dominate in sites with moderate ecological status, which are replaced with species of the Ceratopogonidae, Simuliidae, Tabanidae and Erpobdellidae families in more polluted sites [35].

The basin accommodates many macroinvertebrate cosmopolitan species but also many Balkan or Greek endemics, such as the caddisflies *Hydropsyche peristerica* [37], *Rhyacophila loxias* [38] and *Tinodes alepochori*, the mayflies *Ecdyonurus moreae* and *Ecdyonurus graecus* and the stoneflies *Brachyptera graeca* and *Protonemura intricata taygetiana*, while very recently new gammarid species have been discovered [39]. Apart from the three aforementioned caddisfly species, other common species in the basin are *Rhyacophila palmeni*, *Oxyethira falcata*, *Philopotamus montanus*, *Wormaldia subnigra*, *Polycentropus excisus*, *Tinodes braueri*, *Halesus digitatus* and *Micropterna caesareica*. Regarding mayflies, the most commonly occurring species are *Baetis rhodani*, *Centroptilum luteolum*, *Caenis* sp., *Ecdyonurus graecus*, *Serratella ignita*, *Rhithrogena diaphana*, *Rhithrogena semicolorata* and *Habrophlebia fusca* (Fig. 6).

Regarding damselflies, the most commonly abundant species in the basin are *Calopteryx virgo*, *Platycnemis pennipes* and *Ischnura pumilio*, while *Anax imperator*, *Onychogomphus forcipatus forcipatus*, *Crocothemis erythraea* and *Trithemis annulata* are among the most common dragonflies (Fig. 6). The Plecoptera fauna is mostly represented by *Brachyptera graeca*, *Amphinemura quadrangularis*, *Nemoura flavispana*, *Leuctra hippopus*, *Isoperla tripartita tripartita*, *Eoperla ochracea* and *Perla marginata* [40].

True flies (Diptera) are especially abundant and widely distributed in the basin (Fig. 6), particularly the Chironomidae family, which is very well represented with 43 taxa belonging to five subfamilies (Chironominae, Diamesinae, Orthocladiinae, Prodiamesinae and Tanypodinae). *Brillia bifida* is the most abundant and common species (in terms of the number of sites which it was found) followed by *Polypedilum convictum* type, *Chironomus plumosus* type, *Conchapelopia* sp., *Orthocladius* type S, *Cricotopus bicinctus* and *Rheocricotopus chalybeatus* [41].

Despite the natural and anthropogenically induced drought, the benthic fauna is well adapted to the intermittent character of the river. Differences in species richness variation and macroinvertebrate assemblages between perennial and intermittent streams that are not affected by pollution are not significant [21]. However, the combined effects of drought and pollution exert greater impacts on water

Fig. 6 The Evrotas benthic fauna. From *top left* to *bottom right*: *Ephemera lineata*, *Perla marginata*, *Habroleptoides* sp., *Hydropsyche peristerica*, *Perla marginata*, *Rhyacophila loxias*, *Notonecta maculata*, *Orthetrum brunneum*, *Crocothemis erythraea*, *Hydropsyche peristerica*, *Gammarus* sp.

quality and consequently on macroinvertebrate species; several studies carried out in the ERB have shown that intermittent stream tributaries affected by organic pollution do not recover successfully as perennial ones do, after polluting episodes and the self-purification capacity of intermittent rivers is significantly interrupted [36].

3.3 Fish Fauna

Various hypotheses have been proposed to explain the origin and diversity of endemic freshwater fish in Greece and the Balkan Peninsula in general (for details see [42–44]). Due to the presence of mountainous barriers and deep seas, which were not drained during marine regressions, the southern Greek ichthyofauna includes the most ancient freshwater endemics [29]. This is particularly the case for the ERB, which represents one of the most isolated parts of the Greek peninsula [24]. The river is included in the Ionian ecoregion, having however the most distinctive ichthyofaunal composition in this ecoregion [30, 45]. The river's ichthyofauna comprises ten species, including three of widespread marine origin (restricted only to the lowest most parts of the delta). Compared to other major Ionian ecoregion rivers, it is a species depauperate assemblage. The Evrotas River harbours three unique and range-restricted endemic species of high conservation value listed in IUCN [46], i.e. the Evrotas chub *Squalius keadicus* [47] classified as 'Endangered', exclusively restricted to the Evrotas River and hydrologically connected to the Vasilopotamos River (a spring-fed distributary in the Evrotas Delta plain); the Spartan minnow roach *Tropidophoxinellus spartiaticus* classified as 'Vulnerable' and endemic to the southern Peloponnese Peninsula; and the Laconian minnow *Pelasgus laconicus* [48] classified as 'Critically Endangered', endemic to the Evrotas and upper Alfeios River.

The Evrotas chub (Fig. 7) represents a relic cyprinid species, with remarkable interest for its evolutionary history. In fact, in molecular reconstructions of the

Fig. 7 From *top left*: Peloponnesian endemics – Evrotas chub (*Squalius keadicus*), Spartan minnow roach (*Tropidophoxinellus spartiaticus*) and Evrotas minnow (*Pelasgus laconicus*) – and Mediterranean endemic, freshwater blenny (*Salaria fluviatilis*)

phylogeny of leuciscin cyprinids, the Evrotas chub holds one of the two basal branches in the phylogenetic tree. According to Tsigenopoulos and Karakousis [49], the genetic divergence of the Evrotas chub started in the late Miocene, 5.5 million years ago (mya); however, another view maintains that the divergence started 10.6 mya and coincided with the splitting up of the Aegean Arc [50, 51]. In addition, Perea et al. [52] highlighted the importance for leuciscins of the opening of Aegean Sea (Late Serravallian, 12 mya; Tortonian, 10/9 mya), which resulted in the diversification of some Greek and Anatolian leuciscins. Whatever the case, the Evrotas chub is one of the most ancient European cyprinid species, included in phylogenetic comparisons attempting to elucidate the evolutionary history and taxonomic relations of European cyprinids (e.g. [52]). Its present-day distribution is restricted to Laconia (Evrotas and Vasilopotamos Rivers). However, there is evidence from genetic studies that the historic range of this species was wider and included rivers of the southwestern Peloponnese from which it was extirpated by introgression with new *Squalius* spp. invaders [52, 53].

The Spartan minnow roach (Fig. 7) belongs to the Greek endemic *Tropidophoxinellus* sp., endemic to the Ionian region [29]. Its distribution is confined to the southern Peloponnese (Evrotas, Vasilopotamos, Pamissos, Peristeras and Neda Rivers). Its sister species, *T. hellenicus*, inhabits the Acheloos River in Western Greece and the Pinios River in the Peloponnese. Both species are characterised as fluviolacustrine. It is very likely that the ancestral habitats of the Spartan minnow roach included lacustrine environments, since there is geological evidence that a large lake existed in the middle and lower parts of the Evrotas River in Pliocene times.

Finally, the Laconian minnow (Fig. 7) belongs to the genus *Pelasgus*, endemic to the Balkans. They were earlier placed as *Pseudophoxinus*, which includes numerous species in Asia Minor, Levant, Algeria and Tunisia [54]. Currently, there are seven described species in this genus (*P. epiroticus* (possibly extinct), *P. thesproticus*, *P. marathonicus*, *P. stymphalicus*, *P. minutus* and *P. laconicus*). The Laconian minnow inhabits the Evrotas River and the Vasilopotamos River and has also been found in the springs of Kato Assea (Alfeios River). The rest of the Alfeios River is inhabited by the related species *P. stymphalicus*. Interestingly, ancient writers mention that the Kato Assea springs of the Alfeios R. and the Skortsinou springs of the Evrotas River were hydrologically connected through surface flow. This past connection of the two spring areas may explain the common occurrence of the Laconian minnow in the Evrotas River and the upper portion of the Alfeios River; however the two spring areas are now far apart and separated by mountain barriers.

The native fish fauna of the Evrotas River also includes the European eel (*Anguilla anguilla*) and the peri-Mediterranean river blenny (*Salaria fluviatilis*). In addition, two nonindigenous species have also been recorded in the basin, the eastern mosquitofish (*Gambusia holbrooki*) which has been successfully established in the lower part of the river and the rainbow trout (*Oncorhynchus mykiss*) which was intentionally released for recreational angling in some upper river reaches and has also escaped from aquaculture facilities, however, without establishing a viable population into the river.

Considerable amounts of useful information on the ecology and biology of the river's ichthyofauna are provided mainly from empirical observations, ad hoc research efforts and recent research projects [13, 21]. Table 1 presents the distribution and provenance of the Evrotas fish fauna and summarises some ecological and life history traits.

The Evrotas chub was reported to prefer habitats with slow flow like most *Squalius*. However, Barbieri et al. [55] and Skoulikidis et al. [21, 28] reported that it is a highly energetic and strongly rheophilic species. Recently, Vardakas et al. [56] investigated the early summer habitat use of all three endemic fish species of the Evrotas River and identified the Evrotas chub, which could be characterised as a habitat generalist, since it occupied both slow- and faster-flowing microhabitats. However, data on seasonal habitat use are required, especially during the spawning period (April–May), as a shift to higher velocity areas has been observed in other Mediterranean chub species in spring [57, 58]. The Evrotas chub can reach a maximum size of 25 cm in total length over 5 years. Both, females and males mature at the second year (at about 10 cm SL), and the breeding season takes place in mid-spring (second half of April and beginning of May). The species produces adhesive yellowish eggs, about 2 mm in diameter, from which unpigmented embryos, about 5.1 mm TL, hatch out [55]. The diet of the Evrotas chub consists mainly of insects and various invertebrates; however, small fish cannot yet be excluded.

The Spartan minnow roach is a less rheophilic species than the chub, showing a marked preference for waters with slow flow [56]. It is a strongly phytophilic species that depends on vegetation for reproduction, foraging and protection from natural enemies. It is often found in pools and backwaters, hidden among aquatic plants. It reaches about 15 cm in total length. Age of maturation is suspected to be at their second year for males and females, and it breeds in April and May. Their diet includes insect larvae, invertebrates, molluscs and algae [59].

The Laconian minnow is a stagnophilous (living in marshes) or limnophilous (i.e. springs, lakes, ponds) species, showing little mobility and preferring protected sites in rivers, often with stagnant waters or sluggish flow [56]. This species presents life history traits such as small body size, short life span, rapid growth rate, high fecundity, early maturation and protracted spawning season, which confer a survival advantage in low volumes of water and generally under harsh environmental conditions. The presence of aquatic vegetation is an important habitat requirement for this species. During the reproductive period, it lays down adhesive eggs on aquatic plants. Its food consists of algae and a great variety of small organisms [59].

Past ichthyological research indicated that all native fish species were widely distributed along the entire Evrotas River and its tributaries. However, in the 1980s there was evidence of contraction of the species' ranges and indication that some species were under threat. A severe and prolonged drought between 1989 and 1992, combined with extensive water abstraction for irrigation, caused almost the

Table 1 The distribution, provenance (Native = N, Introduced = I) and the ecological and life history attributes of the Evrotas fish fauna

Family			Ecological and life history attributes			
Scientific name	Distribution	Provenance	Reproductive guild	Flow affinity	Thermal tolerance	Longevity
Cyprinidae						
Squalius keadicus [47]	Evrotas, Vasilopotamos	N	Lithophilic	Rheophilic	Cold water	√
Tropidophoxinellus spartiaticus	Evrotas, streams of S. Peloponnese	N	Phytophilic	Limnophilic	Eurythermal	√
Pelasgus laconicus [48]	Evrotas, Assea Springs (Alfeios)	N	Phytophilic	Limnophilic	Eurythermal	
Salmonidae						
Oncorhynchus mykiss	Widespread	I	Lithophilic	Rheophilic	Cold water	
Blenniidae						
Salaria fluviatilis	Mediterranean	N	Speleophilic	Rheophilic	Eurythermal	
Anguillidae						
Anguilla anguilla	Widespread	N	Pelagophilic	Eurytopic	Eurythermal	√
Poeciliidae						
Gambusia holbrooki	Widespread	I	Ovoviviparous	Limnophilic	Warm water	

complete desiccation of the river and its tributaries. Fish populations were completely extirpated from most of the basin, and only remnant populations remained in few undesiccated sections. However, few years later fish populations managed to become re-established in the river main stem but not at the majority of tributaries [59]. Nowadays, fish assemblages assort at the longitudinal axis of the Evrotas main stem, characterised by a decrease of the Evrotas chub and an increase of the Spartan minnow roach, while the Laconian minnow displays a patchy distribution [60].

3.4 Phytobenthos

Around 100 diatom species have been identified in the Evrotas River, most of them in low abundances. Along the river, the ubiquitous species *Achnanthidium minutissimum* present the highest abundance, and together with other species of the genus (*A. affinis* and *A. pyrenaicum*) constitute up to 90% of the assemblage in the northern parts of the river. In the south, *Achnanthidium* spp. present lower relative abundances, coexisting with species of the genus *Cocconeis*, *Encyonema* and *Encyonopsis*. Around the city of Sparta and its wastewater treatment plant, in the summer, more pollution-tolerant, saprophytic species dominate, such as *A. saprophylum*, *Nitzschia palea* and *Navicula* spp.

The seasonality that exists in the Evrotas River influences both the structure and composition of the diatom assemblages. During spring, the higher flow favours a few species that dominate the assemblage, especially *A. minutissimum*, a species usually dominating in well-mixed, high-quality waters. On the other hand, during summer, the lower flow allows more species to coexist, increasing evenness in the assemblage. This does not necessarily lead to quality degradation, except in the case of sites near Sparta, where an increase in phosphate concentration, possibly due to increased retention of sewage discharge, favours species with higher nutrient requirements. Overall, the water quality in the Evrotas River can be characterised as high to good, based on benthic diatoms.

3.5 Reptiles and Amphibians

While the river delta hosts a diverse and abundant reptile and amphibian fauna, specific amphibian populations, including the toads *Bufo bufo* and *Pseudepidalea viridis* and the frogs *Hyla arborea*, *Pelophylax kurtmuelleri* and *Rana graeca*, *Rana dalmatina*, are scattered throughout the ERB. In addition, water snakes (*Natrix tessellata*), various colubrid snakes and even the sand boa (*Eryx jaculus*) are often seen at the river floodplain and in dried river beds. Tortoises (*Testudo hermanni* and *T. marginata*) and terrapins are quite common, including the widespread Caspian striped-necked terrapin *Mauremys rivulata*, and the less common *Emys orbicularis*, which inhabit the river channels and canals. The Evrotas Delta beaches are important nesting areas for loggerhead turtles (*Caretta caretta*). Many wholly terrestrial reptiles often find food and shelter near the river as well as in the riparian areas. These include the Greek rock lizard *Lacerta graeca*, endemic of the Peloponnese, while the limbless skink *Ophiomorus punctatissimus*, which has a very limited distribution range, has also been recorded in the ERB [61]. This rich reptile fauna benefits by the low-intensity livestock grazing, the small-scale farming and the remarkable habitat variety along the river valley and its tributaries. Land use changes such as intensive agriculture, agrochemical, housing developments and a denser road network may impact the populations.

3.6 Birds and Mammals

Studies of birds have primarily concentrated on the Evrotas Delta, a well-known Important Bird Area [62], serving as a significant wintering and staging area for migratory birds [8]. Although the river valley with its narrow floodplain marshes and riparian woodlands does have some documented interest for a variety of bird species, the delta is outstanding since it holds extensive wet grasslands, reed swamps, brackish marshes, lagoons, beach dunes and river mouth habitats. These habitats are rare in the surrounding areas, and waterfowl, waders, fish-eating birds and birds of prey, many of which are now threatened in Greece, have been recorded (target species for the conservation of the site include *Plegadis falcinellus*, *Falco naumanni*, *Aquila clanga*, *Ixobrychus minutus* and others) [62]. Patterns of bird movement, breeding bird communities and the needs of certain scarce species have been poorly studied in the ERB as a whole.

Mammal assemblages are species poor in this part of the Peloponnese, but these have also been inadequately studied. The water shrew *Neomys anomalus*, the bat *Pipistrellus pipistrellus* and the golden jackal *Canis aureus*, recorded in the ERB, are mentioned in the Greek Red Data Book as 'Insufficiently Known', 'Endangered' and 'Vulnerable', respectively [63]. The populations of golden jackal have plummeted during the last few decades in the southern Peloponnese [64]. A variety of small mammals use the riparian areas of the Evrotas and its tributaries, including widespread and abundant species such as beech marten *Martes foina*, badger *Meles meles* and, least, the weasel *Mustela nivalis galinthias*. The populations of the European otter *Lutra lutra* are especially important in the Evrotas Delta [65]; the species is found along the entire main stem of the river, especially where fish survive. It is interesting to note that the survival of otters during prolonged drought may depend on spring-fed pools and habitat refuges such as in gorges and extensive riparian thickets, since large areas of the basin are nearly totally desiccated during these times.

4 Anthropogenic and Climate Change Pressures and Impacts

4.1 Hydrological Alteration

Until the mid-twentieth century, the economy of the ERB was characterised by slow growth rate. Most of the agricultural land was not irrigated, and the extraction of substantial surface and subsurface water resources was maintained in low levels [3, 17]. As indicated in [21], land reclamation and irrigation works helped to expand the production and variety of irrigated cultivations, particularly the promotion of citrus plantations after World War II. Up until the 1950s, the greatest part of the Evrotas watercourse maintained near-natural hydromorphological features and sustained surface water throughout the year, which is also supported by the former existence of fish in many of the Evrotas' tributaries [20, 59]. However, during the period between the 1950s and 1990s, large-scale irrigation, river engineering and drainage projects were carried out, targeted at flood retention, control of diseases and irrigation of agricultural land. In the last three decades, the ERB has evidenced rapid hydromorphological alterations as irrigated cultivations expanded towards natural and seminatural land including riparian areas, while natural vegetation along the riparian zones has been dramatically altered while, more recently, the irrigation network expanded to olive groves [9, 66]. This surface and groundwater over-exploitation (Fig. 8), especially during the dry years, affected the basin's hydrological regime and resulted in the lowering of groundwater tables [28]. During extreme drought periods, such as between 1989 and 1993 and 2007 and 2008, increased water requirements for irrigation resulted in the depletion of groundwater aquifers and, consequently, in the desiccation of extensive parts of the river network. These practices resulted in an artificial desiccation of the river network during the summer periods and significant morphological degradation of the river system, limiting water and habitat availability and severely affecting the aquatic and riparian biota. The increase of pollution effects during drought years is also evident [24, 67, 68].

Quantification of the effects of agricultural water exploitation on the Evrotas flow regime was applied by Skoulikidis et al. [21]. In this study, the water balance at the ERB was estimated, according to the current water uses, and the current water uses excluding the water used for irrigation of olive groves. The decision to adopt this approach was based largely on the fact that irrigation of olive trees has only recently been introduced into the basin. The results of the monthly water budget analysis indicated that at the end of the hydrologic period the remaining water resources in the catchment were less than 2.9 m^3/s. Considering that the total irrigation requirements prior to the irrigation of olive groves (77% of the ERB's agricultural land) were 30% of the current uses, the discharge at the river outflow during the dry period was estimated at 9.4 m^3/s, which is over 3 times higher than the current one. Figure 9 reveals that the water quantity used for olive grove irrigation alone would be enough to maintain river flow during the dry season.

Fig. 8 Water abstraction for irrigation at the lower parts of the river. Eutrophication is also evident

According to meteorological data, every 7–8 years, the ERB has been affected by severe droughts that reached a peak during the end of 1980s to the beginning of 1990s. Other severe droughts took place in 1977, between the years 1983 and 1986, in 2004 and finally between 2006 and 2008, while 2007 had extremely low rainfall and snow cover was also limited [21].

Within the last 35 years, a long-term decrease of both rainfall and discharge is evident in the ERB. However, a 10.5% long-term (1974–2008) rainfall decrease was accompanied by a 51.2% diminishing of river discharge for the same time period (Fig. 10), and the estimated rainfall elasticity to stream flow, calculated for

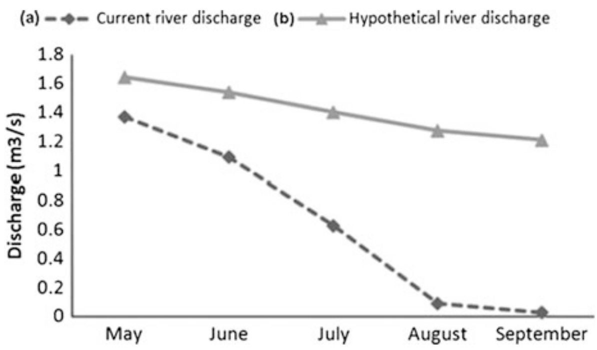


Fig. 9 Monthly discharge variation during the irrigation period of the Evrotas River at Vrontamas for two different water abstraction states: (**a**) measured average river discharge for the period 1998–2000 and (**b**) hypothetical river discharge without the intense irrigation scheme conditions (source: [21])

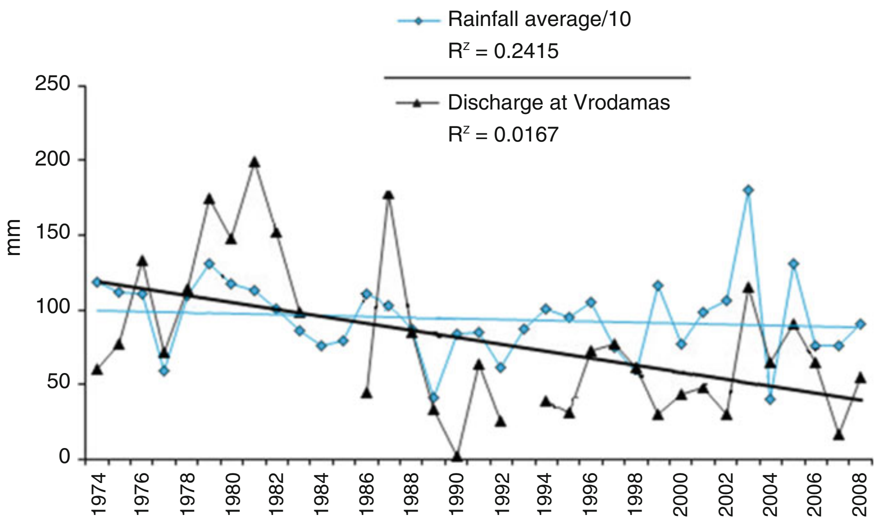


Fig. 10 Long-term (1974–2008) variation of average rainfall and discharge at Vrontamas. *Bold line* presents the 5-year moving average of discharge

the same period, was very low compared to other catchments elsewhere in the world [21]. The decrease of discharge is the highest among 15 major Balkan rivers [24]. These results highlight the impact of water abstractions on the river's hydrological regime.

Extreme drought events strongly disturb the flow regime throughout most of the river, as was documented in 2007. During April 2007, there was flow in most of the river network. By the end of September to mid-October, water had vanished from almost all tributaries and most segments along the main river course were completely desiccated (Fig. 11). Only some spring-fed river reaches (comprising 20% of the main river's length) and a small number of headwater tributaries retained surface water. The hydrological condition of the river during summer and autumn 2008 was quite similar to that of summer/autumn 2007. With progressing desiccation, isolated remaining pools of various sizes were formed in several main reaches and tributary sections. A number of pools dried out completely resulting to (the) massive fishkills. Some of the largest pools persisted throughout the summer, providing shelter for fish and other aquatic organisms. However, pool depth and surface area decreased constantly, resulting in severe crowding of fish populations.

Historical records of fish occurrence and interviews with local people indicate that in the past, all fish species were widely distributed throughout the entire river and its tributaries. Due to human interventions (mainly severe water abstraction but also pollution and morphological alterations), fish populations are constantly declining. This is especially true in the tributaries of the Evrotas River, where summer drying is now a seasonally predictable event and most fish populations are permanently extinct.

The macroinvertebrate fauna of the Evrotas shows high resilience to drought, as many species recover successfully when flow returns [21]. In fact, post-drought macroinvertebrate community was relatively similar to the pre-drought one (approx. 70% similarity), and minimum differences in assemblages were mostly attributed to seasonality. The community structure of intermittent streams was similar to that of permanent streams, and no marked differences were observed

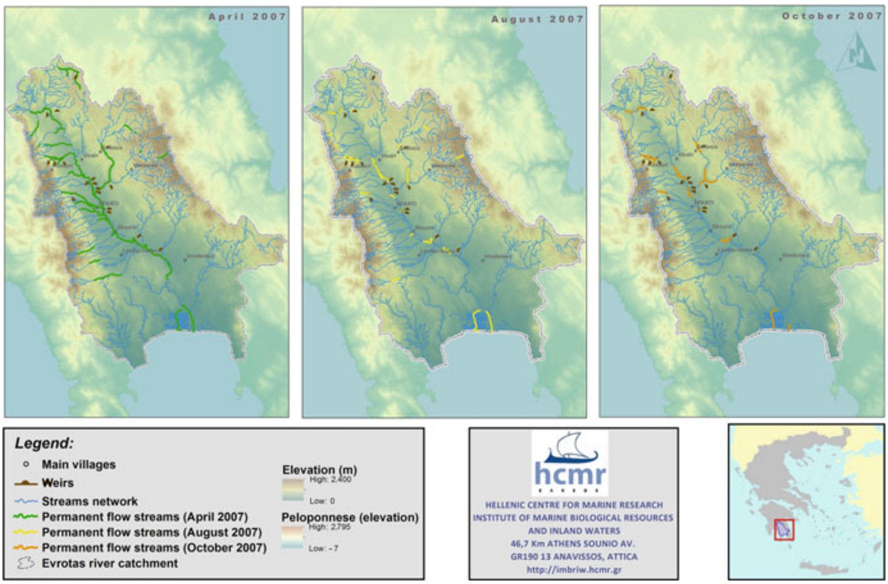


Fig. 11 The active hydrographic network during the wet (spring 2007), dry (August 2007) and end of the dry season (summer 2007)

between macroinvertebrate assemblages. Although temporal features may decrease or even eliminate species richness, as well as species size, summer drying does not seem to affect the recolonisation potential of the Evrotas benthic fauna [21]. However, since historical data are lacking, it cannot be excluded that repeated and prolonged droughts during the past years have already changed macroinvertebrate communities towards species more resistant to droughts.

4.2 Point Source and Diffuse Pollution

Point source pollution in the ERB comes primarily from municipal and industrial wastewaters. Almost 63,000 inhabitants from 95 municipal districts, including Sparta, the capital city of the Laconia Prefecture, are recorded in the basin area. The ERB has only one wastewater treatment plant (WWTP) in the city of Sparta, serving the local population. The chemical properties of the WWTP's are $BOD_5 = 6$–14 mg/L, COD = 12–43 mg/L, $N\text{-}NH_3^+ = 1.5$–1.8 mg/L, $N\text{-}NO_3^- = 1.4$–4.3 mg/L and total $P = 2.6$–4.2 mg/L [69]. All settlements outside the city are served by private permeable and impermeable cesspools. Domestic wastewater from permeable cesspools gradually infiltrate to the groundwater aquifers, while those from impermeable cesspools are often discharged illegally into the river, often resulting in severe eutrophication at the downstream parts of the river course (Fig. 12). Industrial pollution comes from two fruit juice production companies, a meat-processing factory, almost 60 olive mills, specific large livestock breeding units and olive

Fig. 12 Eutrophication downstream of Sparta near Skoura village

treatment-standardisation units. In addition, oil core plants and cheese manufacturing units operate in the basin.

The chemical properties of olive mill wastewaters (OMW) have been well documented, having very low pH and being extremely rich in nitrogen, phosphorus and phenolic compounds, which are toxic to aquatic organisms [70–72]. The operation of olive mills is seasonal and usually lasts between 3 and 5 months (November–March) where large amounts of OMW are annually produced.

In 1998, the annual amount of BOD_5, COD and total phenols from OMW in the river basin was estimated at 4,430 tons, 9,080 tons and 1,040 tons, respectively [73]. Although supposedly olive mills deposit their wastewater into the basin's streams and rivers after specific treatment, unprocessed OMW is being discharged regularly throughout the river basin. In addition, effluents from the juice production unit are occasionally discharged in the basin without any treatment.

Recently, the effects of OMW on the stream macro-invertebrates, water quality and river ecological status were thoroughly and systematically studied in the Evrotas River Basin [35, 36, 71]. The results of these studies revealed the spatial and temporal structural deterioration of the aquatic community due to OMW discharge with consequent reduction of the river capacity for reducing the effects of polluting substances through internal mechanisms of self-purification. OMW, even highly diluted, had significant impacts on the aquatic fauna and the ecological status of the Evrotas River. The vast majority of macroinvertebrate taxa were eliminated, and only a few tolerant Diptera species (i.e. Chironomidae, Simuliidae, Syrphidae) survived with very limited abundances (1–4 individuals/1.25 m^2). Macroinvertebrate assemblages downstream the OMW outlets were dominated by Diptera species, whereas Ephemeroptera, Plecoptera and Trichoptera (EPT), which are pollution intolerant, were almost depleted during and after the OMW discharge period (Fig. 13).

The ecological status of mountainous sites upstream of olive mills varied from good to high during all months, while minimal variations among and within sites were mainly attributed to seasonality. In contrast, sites downstream of olive mills varied between good and high, before the wastewater discharge period, to moderate and bad during the discharge period. Sites with relatively high slope, altitude and oxygen, presented moderate ecological status due to the high self-purification capacity, whereas sites located in lowlands were classified from moderate to bad [36].

Similar results were also observed with orange juice processing wastewaters (OJPW). Ecological quality monitoring and assessment carried out in these two streams revealed significant loss of the benthic fauna, since during almost every month of monitoring, only a few individuals of the dipteran families of Chironomidae and Simuliidae were found [35]. Diffuse pollution in the ERB includes surface runoff from the urban and industrial areas and the basin's road network, runoff from livestock breeding activities and from fertilisers/pesticides used in agriculture. It has been estimated that 24,641 tons of nitrogen and 9,771 tons of phosphorous are used in the basin (from agriculture, livestock, wet and dry deposition, domestic wastewater and various point sources) [13].

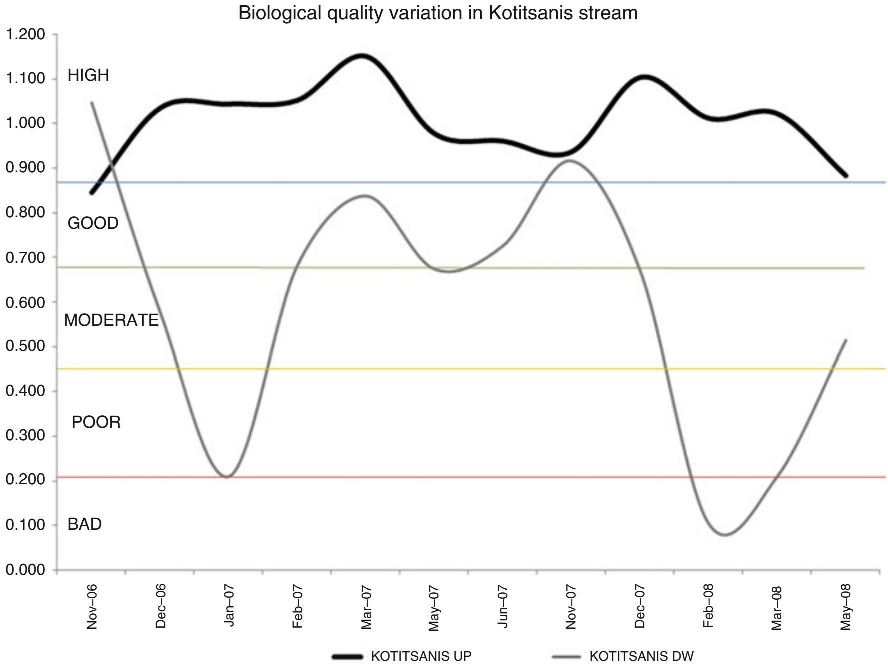


Fig. 13 Variation of biological quality in Kotitsanis tributary upstream and downstream the OMW outlet during 2 years of monitoring

4.3 Forest Fires

Forest fires are usual phenomenon in the ERB due to the semiarid environment, resulting in extended periods of water scarcity, mainly during the summer months. While forest fires are usually restricted, in August 2007, 216 km^2 of forested areas, pastures and olive groves were incinerated, provoking large floods during autumn, in which huge quantities of sediment were transferred downstream. Cropland was reduced by 4.5%, and forests were reduced by 11.8% during these fires.

Research carried out after the catastrophic event showed that the wild fires altered the soil's hydrological and geochemical properties rendering the soil surface more susceptible to erosion. Initial flood events enhanced runoff and erosion rates in burned terrains, causing elevated sediment and dissolved and particulate phosphorus yields. While dissolved phosphorus concentrations where over 20 times enriched against previous levels, particulate phosphorus represented 99% of the burned hillslope phosphorus yield, and up to 20% of total particulate phosphorus in burned sediment was shown to be potentially bioavailable [74].

Overall, taxa richness and total abundance were lower in burned than in reference streams [75]. Specific taxa responded differently to the effects of fire. For example, densities of disturbance-adapted forms (e.g. Chironomidae, *Baetis* spp.)

increased after the fire, while many other taxa showed the opposite response. Adverse effects of wildfire on the biotic community were largely the result of physical changes in habitat due to increased runoff and suspended matter and clogging of river sediment pore space by ash and, possibly, to an over tenfold increase of ammonium shortly after the fire compared to pre- and postfire periods [75]. The ecological status of the sites prior to the wildfire was classified as high, whereas after the wildfire the status declined to moderate.

5 Hydrogeochemistry, Pollution and Biogeochemical Processes

5.1 Major Ions, Physicochemical Parameters and Nutrients

The vast majority of the Evrotas hydrographical network belongs to the most representative hydrochemical river type found in Greece (Ca > Mg > Na > K–HCO3 > SO4 > Cl; [76]), mainly resulting from the dissolution of carbonate rocks. Situated within the southern part of geographical zone No. 3, the ERB presents an example of the influence of dry climatic conditions on river hydrochemistry; in contrast to the rivers located at the northern part of this zone, the Evrotas River is characterised as very hard (median total hardness 307 mg/L $CaCO_3$) and highly mineralised (median total dissolved ion concentration 491 mg/L). As expected, maximum solute concentration occurs during the dry period as a result of the (1) low dilution capacity of the river water due to the lack of rainfalls, (2) increased evapotranspiration and (3) contribution of alluvial aquifers, with higher solute concentrations than the river. Due to the substantial karstic spring inputs, higher hydrogen carbonate concentrations along the Evrotas main stem were found upstream, at the midway and near the estuary. On the contrary, sodium revealed a downstream increase as a result of (1) soil salinisation processes (due to irrigation of agricultural land) (Fig. 14), (2) impacts of olive mill wastewater discharge which are rich in salts [36], (3) salinisation of coastal aquifers and (4) transport of sea salt aerosol.

In general, the Evrotas River presents satisfactory oxygenation conditions [21]. However, the illegal discharge of domestic wastewaters, as well as the malfunction of the WWTP in Sparta, may result to anoxic conditions. In fact, automatic monitoring revealed recurrent zero drops of dissolved oxygen lasting for several consecutive days during the summer months of 2009 downstream of the WWTP (Fig. 15) [77]. Anoxic conditions are accompanied by low pH levels suggesting increased respiration of organic matter. In addition, eutrophication is a common phenomenon in summer especially downstream of Sparta (Fig. 12). In the vast majority of the examined area, the molar ratio between nitrogen and phosphorus is by far more than 16 (mean: 67), thus indicating phosphorus-limited

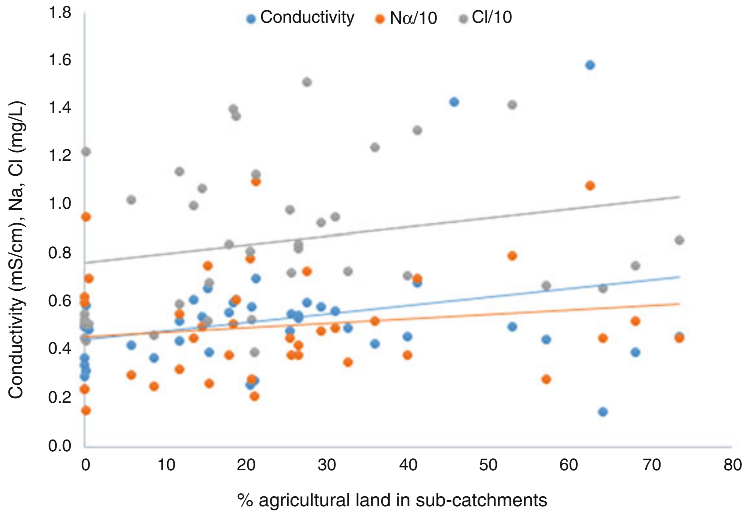


Fig. 14 Correlation between conductivity and the concentrations of sodium and chloride vs agricultural land percentage in the examined sub-catchments

photosynthesis. Hence, to control eutrophication, management plans should focus on phosphorus reduction [28].

Phosphorus levels in the Evrotas River are considered low (mean P-PO_4: 16 μg/L, mean TP: 40 μg/L), assessed as 5 times lower than the European average (78.8 μg/L, according to http://www.eea.europa.eu/soer-2015/countries-comparison/freshwater, accessed March 2016). Low phosphorus concentrations have also been reported for other Greek carbonate basins located in zone No. 3, probably due to adsorption mechanisms on carbonate material [24, 78]. Nitrate concentration has been positively correlated with the percentage of agricultural land in the respective subbasins (Fig. 16), suggesting the presence of nitrogen fertilisers. However, the mean nitrate concentration in the ERB (0.55 mg/L N-NO_3) was found to be 4 times lower than the mean nitrate concentration found in other European rivers (2.2 mg/L N-NO_3, according to [79], data 2010).

Riverine biogeochemical processes intensify due to the increasing impact of point source pollution, such as agro-industrial and domestic wastewaters and WWTP effluents, as well as due to the diminishing water volume during summer droughts and the initial autumn flash floods [14]. Research has shown that with increasing desiccation, intensification of photosynthesis was apparent. This resulted in nitrate and silicate assimilation and carbonate precipitation, even in unpolluted river reaches. Carbonate precipitation is particularly pronounced in intermittent reaches, where surface precipitates on riverbed pebbles and cobbles were investigated [80]. As lentic conditions establish, and water temperature increases, respiration processes become more dominant (Fig. 17). With increasing lentification, a decrease of nitrate accompanied with a rise in ammonium may be attributed to nitrate reduction [81]. A reduction of total nitrogen (TN) in isolated pools may

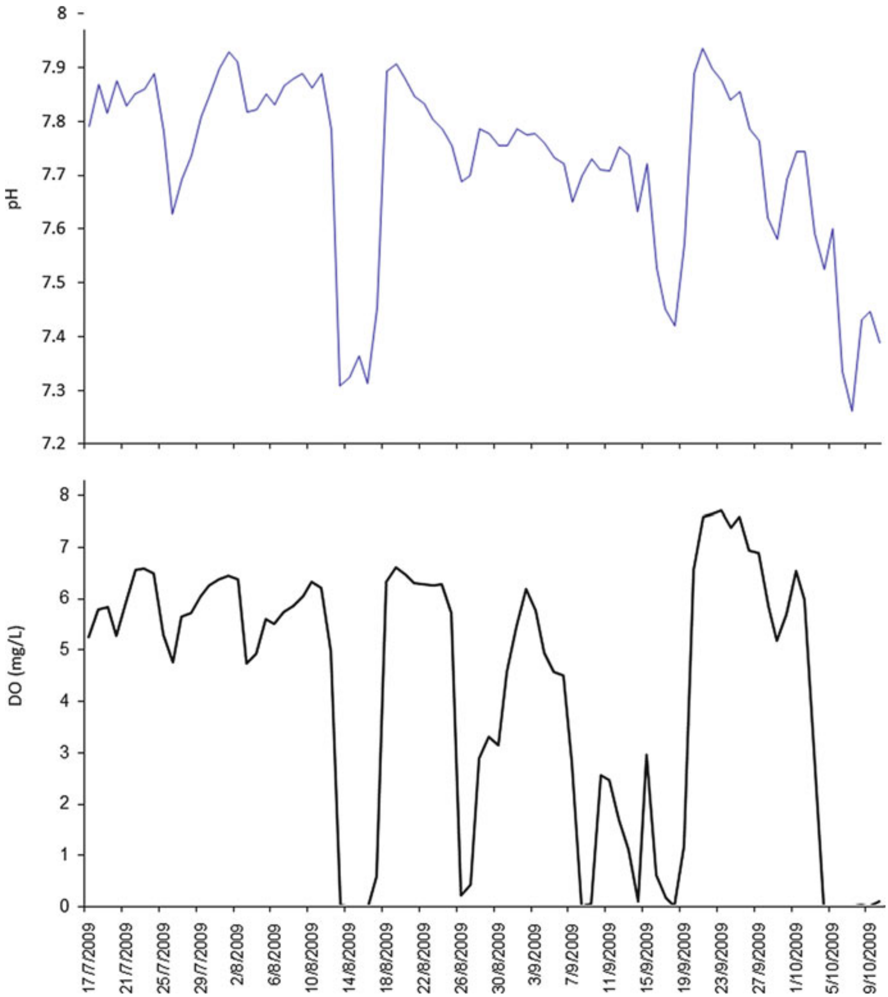


Fig. 15 Fluctuation of daily average dissolved oxygen (DO) and pH levels (derived from 10-min automatic measurements) downstream of the inflow of the Sparta-WWTP effluents

result from denitrification and/or ANAMMOX (anaerobic ammonium oxidation) [82] processes associated with benthic sediments colonised by microalgae [83]. However, the expression of opposing processes in the same water body as a result of differing microhabitat characteristics is not to be excluded [84]. During complete desiccation, processes slow down due to low microbial activity [85–87]. Upon rewetting, these areas comprise biogeochemical 'hotspots' and 'hot moments' [88]. In Evrotas, initial floods caused a rise in nitrate (Fig. 18), nitrite, ammonium, TN and silicate, compared to their annual averages [84]. The increase of nitrogen species and especially nitrate and nitrite, during the flood events, resulted in a decrease of nutrient quality (from moderate to poor for nitrate and

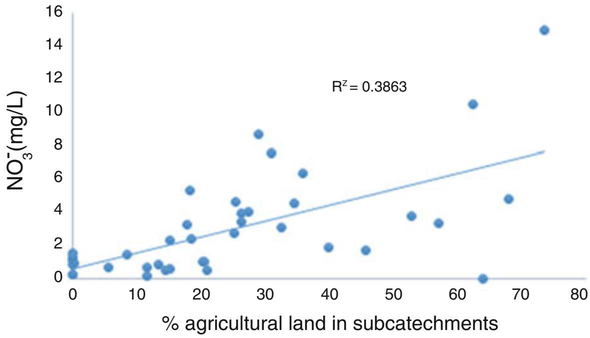


Fig. 16 Correlation between nitrate concentration and agricultural land percentage in the respective sub-catchments

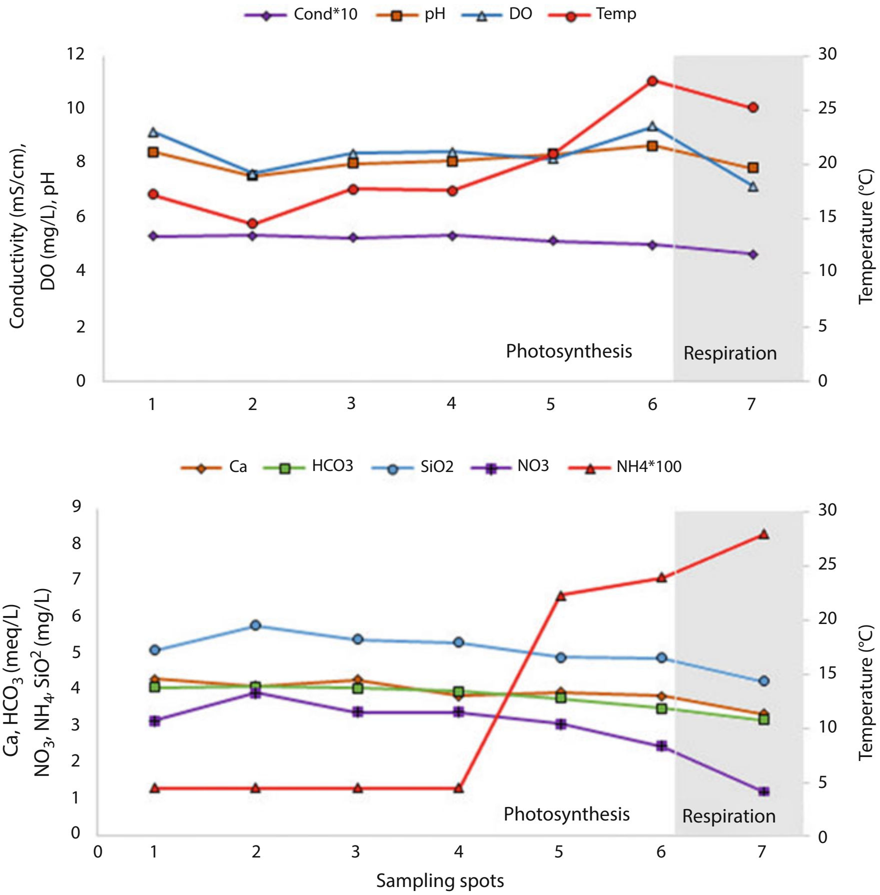


Fig. 17 Major ion and nutrient variation and related biogeochemical processes with increasing spatial lentification (from sampling spots 1–7) during June 2010, at the Sentenikos intermittent reach

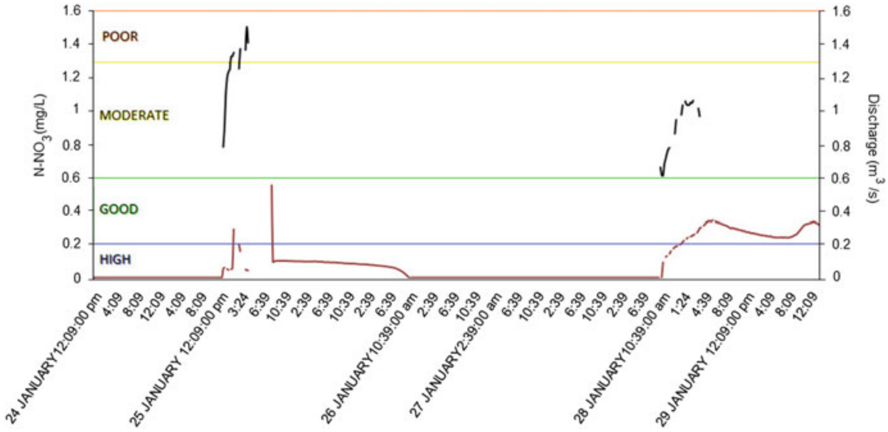


Fig. 18 Nitrate variation (*black line*) during two succeeding initial flood events (*red line* represents discharge variation) that occurred on 25 and 28/29 January 2011, at the Sentenikos Reach. Colours represent nitrate quality classes (*blue* high, *green* good, *yellow* moderate and *orange* poor quality) according to the Greek Nutrient Classification System [76]. *Dotted line* represents the mean annual N-NO_3 concentration

from high to moderate for ammonium) and is attributed to the release of nutrients due to rewetting as a result of osmolysis of soil microbial biomass [89] and, shortly after, to mineralisation and subsequent nitrification of labile organic matter accumulated during the dry season in riparian soils and river bed sediments [90, 91]. Initial floods also caused substantial sediment mobilisation and flushing of epsomite-type salts, which are very soluble [84].

5.2 Micropollutants and Priority Substances

Sediment samples collected within the GLOBAQUA research project (http://www.globaqua-project.eu) in June 2014 (unpublished results) from four reaches distributed throughout the basin, revealed the presence of polar pesticides, halogenated and organophosphorus flame retardants (low concentrations) and several pyrethroids. Regarding pesticides, low concentrations were detected in the eight water samples collected. Only triazine pesticides were detected from the mid- and upper reaches of the river (Koliniatiko and Vivari), while diuron was also detected (104 ng/L) in Vivari. Sediment samples from the same reaches detected concentrations of metolachlor, diuron, alachlor, triazines and organophosphate/carbamates. Metolachlor was the compound found at the highest concentrations, ranging from 0.77 to 38.58 ng/L, with the higher concentration found downstream of the wastewater treatment plant of Sparta. Diuron concentration ranged from 70 to 140 ng kg/L, triazine concentration ranged from 70 to 180 ng kg/L and traces of organophosphates and carbamates from 10 to 30 ng kg/L.

An older study conducted in 12 sites across 6 tributaries of the river, on a seasonal basis from January 2007 to March 2008, revealed the presence of the herbicides metolachlor and alachlor, the fungicides triadimenol and penconazole and the insecticides dimethoate, monocrotophos, malathion, fenthion and carbophenothion [68]. Most pesticides and the highest concentrations were detected in November 2007 and March 2008, above the acceptable limits for potable water (0.1 μg/L), while pesticide concentrations in stream sediments were also significantly high. Alachlor and dimethoate were detected only in the water samples, whereas monocrotophos and carbophenothion were detected only in the sediment samples.

In the same study [35, 68], trace metals in stream sediments were analysed on a seasonal basis (March 2007, November 2007 and February 2008). Trace metals were found in the following decreasing order: Ba > Cr > Zn > Ni > Cu > Pb > As > Mo. Concentrations of barium (Ba) were above the acceptable limits (limits established by various scientific groups and organisations, see p. 3072 in [68]). However, high Ba levels are generally normal, also found in other parts of Greece, and thus are not considered to be of anthropogenic origin. Furthermore, Cr and Ni concentrations exceeded the acceptable limits. These high concentrations, however, occurred at sites with good or high ecological status at relatively high altitudes and in forested areas, suggesting inputs from ultramafic rocks in the region.

Analysis of other micropollutants conducted in 2015 within the framework of the GLOBAQUA project revealed no polycyclic aromatic hydrocarbons (PAHs) concentrations in river water samples. Total levels of PAHs in sediments were always below 1 μg/L, with concentration values up to 70 ng/L. Similar outcomes were observed regarding semi-volatile organochlorine compounds hexachlorobenzene, gamma-hexacyclohexane, *p*,*p*′-dichloro-diphenyl-trichloroethane (DDT) and related compounds (DDX). Hexachlorobenzene (HCB) and gamma-hexachlorocyclohexane (also known as lindane) were not detected in any water sample, and the levels of DDT and DDX were very low and never surpassed the annual averages (AA-EQS – 25 ng/L) established in the Directive 2013/39/EU [92].

Regarding pharmaceutical compounds, 14 out of the 90 analysed compounds were detected. The most frequently detected anti-inflammatory/analgesics were naproxen, ketoprofen, salicylic acid, ibuprofen and diuretic hydrochlorothiazide. Antihypertensive compounds detected were valsartan and irbesartan and the antibiotic azithromycin. The highest concentrations were found downstream of the domestic wastewater treatment plant of Sparta. Pyrethroids, due to their hydrophobic behaviour (Kow between 4 and 7), are not usually found in water. Only permethrin (1.0 ng/L) and tetramethrin (<2.6 ng/L) were detected at the upper reaches of the river, at the Koliniatiko site. The same compound was detected but not quantified in the mid-reaches of the river at the Vivari site (<0.8 ng/L). No other pyrethroid was detected in the water samples. On the contrary, pyrethroids are more likely to be found in sediments. Tetramethrin was found below the MLOQ (method limit of quantitation) in sediments of Koliniatiko and WWTP sites; fenvalerate was detected at the same sites below the MLOQ and 0.02 ng/g dw, respectively. Permethrin, cypermethrin and deltamethrin were found in 100% of the samples, the two latter being the main contributors in all sediments. Total pyrethroid levels ranged from 0.12 to 14.0 ng/g dw. Total levels ranged from 5 to 13 ng/g dw for the Koliniatiko and WWTP sites.

5.3 The Ecological Status of the Evrotas River

The ecological status of the European Union's fluvial ecosystems is assessed within the requirements of the Water Framework Directive 2000/60/EC (WFD). Sampling sites and water bodies are classified using a five-class system (high, good, moderate, poor, bad), which combines physicochemical, hydromorphological and biological data [93], appropriately collected to comply with the requirements of the WFD. In Greece, the physicochemical quality is classified within the Nutrient Classification System [76] and hydromorphological assessment within the River Habitat Survey (RHS) method [94]. The biological quality classification is derived from the lowest observed quality class among the four biological quality elements (BQEs): fish, benthic macro-invertebrates, diatoms and macrophytes. The quality of each BQE is determined independently, according to specific biotic indices [95–99].

The ecological status of the Evrotas River was assessed during 2006–2007 [28] and 2009–2010 and has continued to be regularly assessed since 2013 within the National WFD monitoring programme and since 2014 within the context of the GLOBAQUA project. Physicochemical and biological data have since seasonally been recorded, while hydromorphological assessment to quantify the degree of hydromorphological alteration is applied once every 6 years. While the initial evaluation of 2006–2007 and 2009–2010 included an extended monitoring network of more than 50 sampling sites throughout the basin (Figs. 19 and 20), during the WFD monitoring programme, seven key sampling sites were retained, allocated at the upper parts of the river, at the mid-course and at the most downstream parts including the river delta, in order to cost-effectively describe the gradient of anthropogenic pressures in the ERB. To allow for compatibility and comparability between the sampling periods, comparison is restricted to the seven sites of the WFD monitoring programme (Figs. 21 and 22).

The ecological status assessment carried out during 2006–2007 and 2009–2010, in more than 50 sites of the ERB (Figs. 19 and 20), showed significant degradation in the sites around the city of Sparta, where most industrial activities occur, as well as downstream of the city. In addition, the ecological status of many sites in semi-mountainous areas was assessed as 'good' due to local point pollution sources (e.g. olive mills) and due to hydrological disturbances, which resulted to poor or bad biological status.

The ecological status of the seven monitoring sites of the Evrotas River for the periods 2006–2007, 2009–2010 and 2013 (excluding the highly variable fish fauna) is depicted in Fig. 21. The results reveal a stable, high or good ecological status for the upstream sites (1, 2 and 3), which are less impacted by the previously mentioned anthropogenic pressures. The status of site 4 was assessed as moderate during all

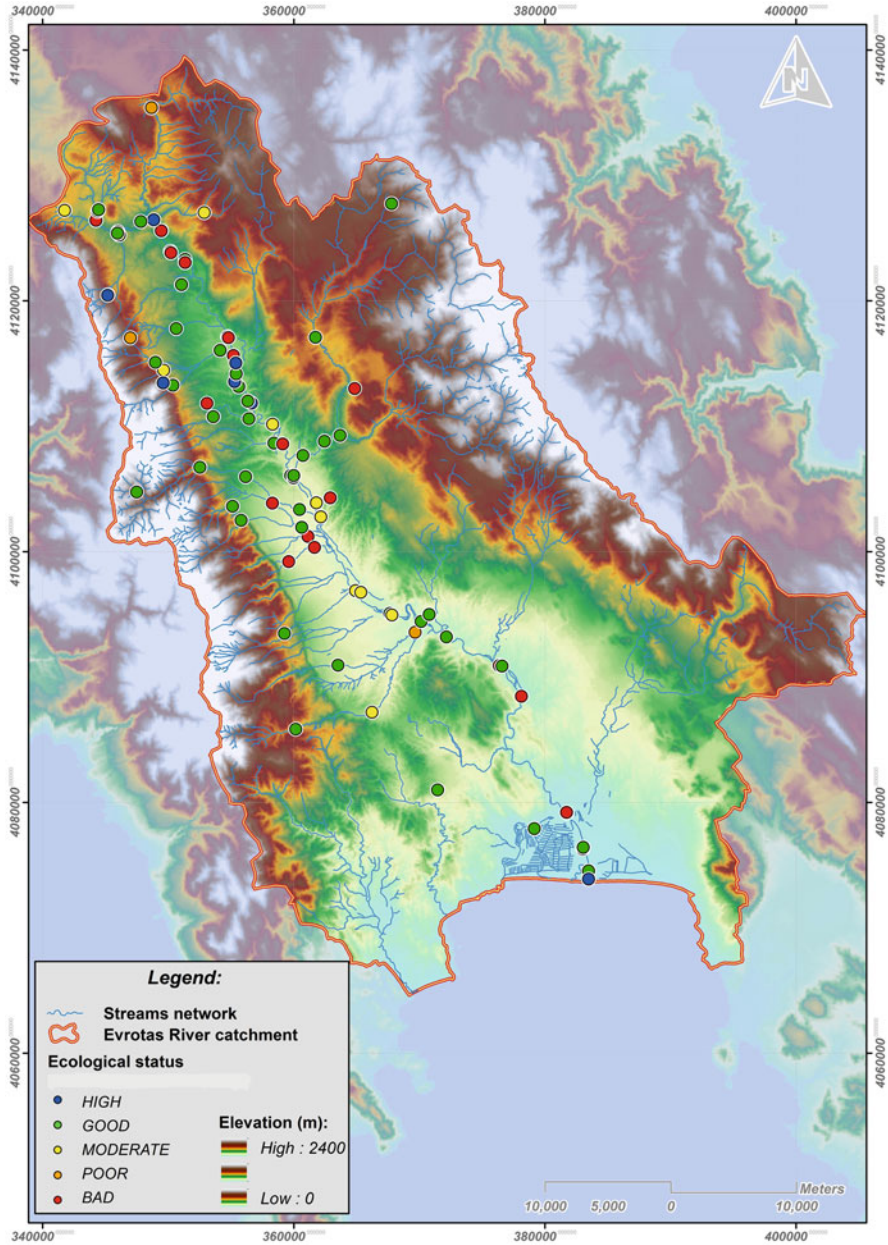


Fig. 19 Ecological status of the Evrotas basin during the extended sampling campaign of 2006–2007

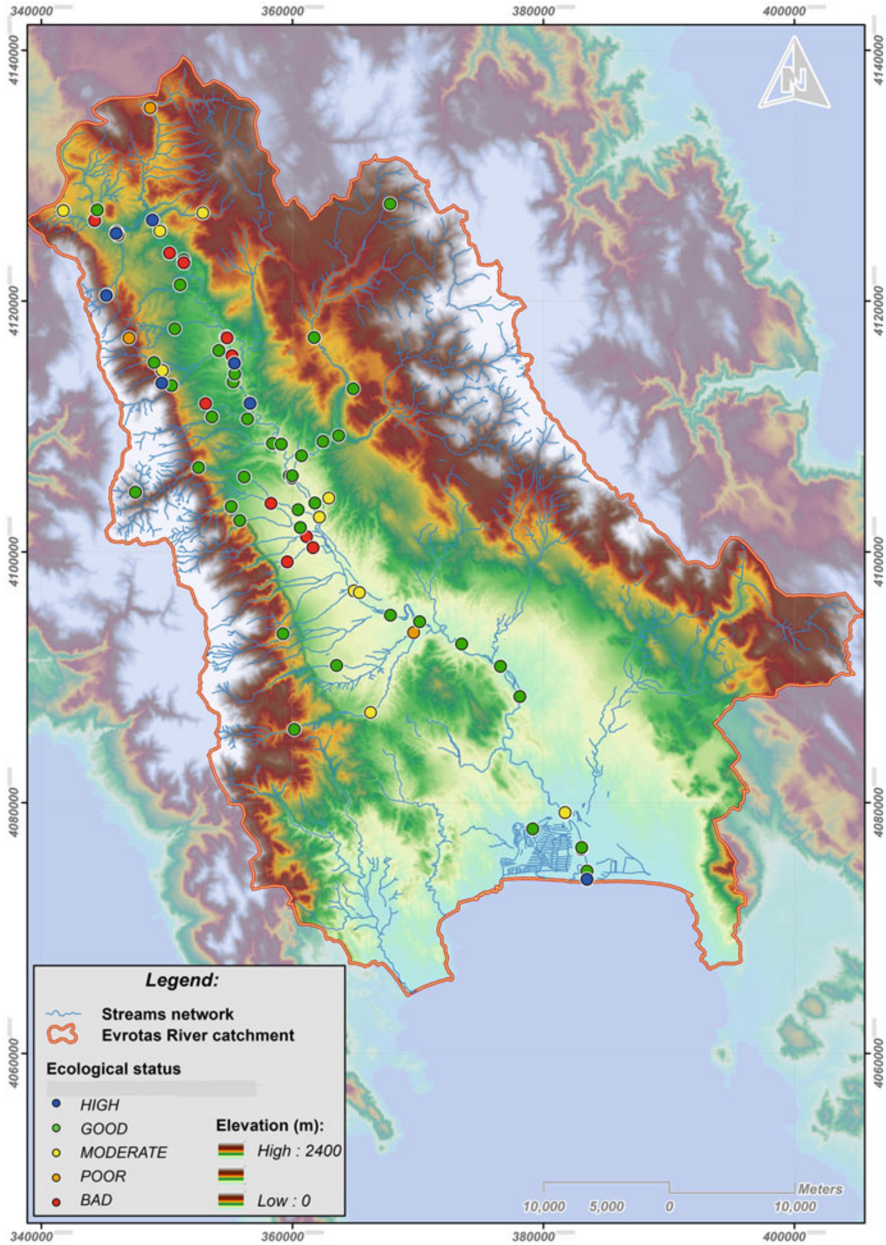


Fig. 20 Ecological status of the Evrotas basin during the extended sampling campaign of 2009–2010

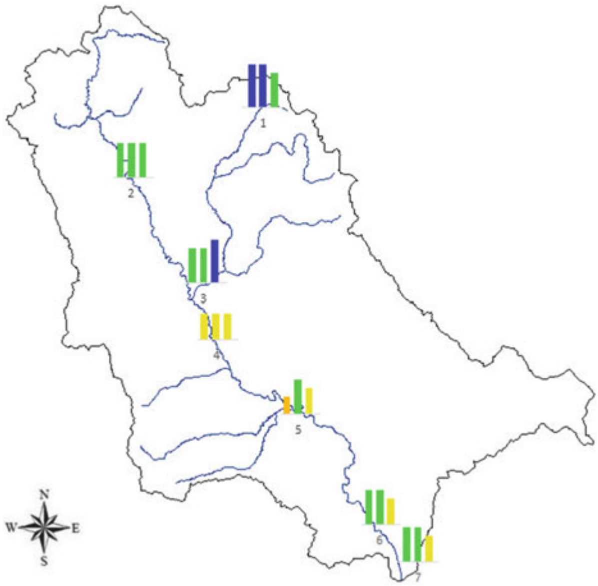


Fig. 21 The ecological status of the sites investigated during the three sampling periods (from left to right: 2006–2007, 2009–2010, 2013). Status colour codes: *blue* high, *green* good, *yellow* moderate, *orange* poor

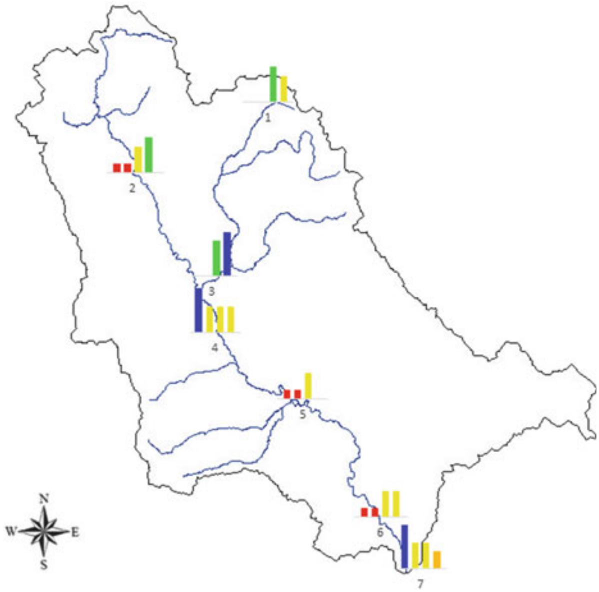


Fig. 22 The fish-based biological quality of the sites investigated during four sampling periods (from left to right: 2007, 2008, 2010 and 2013). Status colour codes: *blue* high, *green* good, *yellow* moderate, *orange* poor, *red* bad

studied periods, possibly reflecting the impacts of wastewater discharge from the treatment plant of Sparta, in combination with the wastewaters from the juice manufacturing factories in the nearby area. Site 5 shows high status variability, ranging from poor to good (Fig. 21). This variation is mainly attributed to the water volume availability that dilutes wastewater loads from the WWTP, olive mills, orange juice production units and agrochemicals. The status of the most downstream sites was degraded from good to moderate, and as a result, in 2013, the mid- and lower reaches of the ERB did not fulfil the requirements of the WFD 2000/60/EC.

The results assembled from the 10-year monitoring and assessment of the ERB, suggest that, overall, the ecological status is moderate, despite the numerous anthropogenic point and diffuse pollution sources and the seasonally severe effects of water abstraction. A declining upstream-downstream trend is indicated, as the human-induced pressures become intensified after the city of Sparta until the river delta. Nevertheless, almost 50% of the watercourse failed to fulfil the demands of the WFD, and measures are essential to reverse this situation. It must be noted that the fish fauna, which is more sensitive to water level fluctuations in the river, was critically impacted by the extreme drought event of 2007 in the ERB as mentioned previously (Fig. 22). This resulted in bad status for sites 2, 5 and 6 during 2007 and 2008. In 2013, fish assemblages were recovered; however, the quality improved only to moderate for all sites except for 2 and 3, which presented high and good status, respectively. Thus, the integration of the fish fauna in the ecological assessment indicates that in 2013, 72% of the watercourse failed to meet the requirements of the WFD.

6 Management and Conservation: Main Actions to Preserve and Restore the Aquatic Ecosystem

Effective river basin management should incorporate scientific information (data from the WFD monitoring programmes) and long-term ecological studies regarding the ecological and chemical status of the river, as well as the opinions, interests and possible conflicts of all stakeholders and managers, including the local communities near the basin. Specifically, the strategic objective is the integrated management of water resources in the Evrotas River Basin that will contribute to environmental status upgrade, social cohesion, local economic added value and improvement in quality of life [13]. Within this optimistic vision, a strategic management plan (SMP) was issued in 2009 [13] for the ERB within the LIFE-EnviFriendly project, incorporating selected measures required to reach the abovementioned generic target. In addition, measures and proposals derived from the GLOBAQUA research project, which succeeded the LIFE-EnviFriendly project, are included which, in turn, enhance the SMP of 2009. Overall, river basin management for the Evrotas should focus on key thematic areas, which are briefly summarised below:

1. Agriculture and water for irrigation
2. Reduction of the point and diffuse pollution sources

3. Floods/droughts prevention measures
4. Biodiversity conservation and restoration of the aquatic ecosystem

6.1 Agriculture and Water for Irrigation

Land used for farming should be of high priority within a sustainable agricultural approach. The missing organic carbon and significant trace elements must be 'returned' to the soil by rotating crops throughout the year, instead of monoculture farming. Crop rotation will enable to replenish the soil with nutrients without the use of fertilisers, while keeping the production cost low. Furthermore, by treatment and utilisation of municipal wastes through the minimisation of landfilling and the maximisation of recycling and composting, sufficient quantities of compost could be produced to enrich the crop fields.

As mentioned previously, there are numerous private and municipal borehole drillings (estimated at around 3,500) for irrigational uses scattered throughout the basin. Irrigation withdrawals were estimated to 62 mm^3 from groundwater wells and 15 mm^3 from direct abstractions from the river [100]. Water from these drillings is pumped from the surface and groundwater network of the ERB without any proper control by a supervising authority. In addition, it is currently very difficult to estimate the real consumption of water; as in many private wells, there are no records concerning well yield or well depth.

The sustainable scheme, however, would require the construction of drip irrigation systems to minimise the water loss and the pricing of the irrigating water according to the actual quantity of the water used and not according to the irrigated area. Information systems could also be established to alarm the water users (farmers), when water reserves are reaching minimum limits, in order to adapt irrigational practices accordingly. Water from the domestic wastewater treatment plant and the agro-industrial facilities is not currently reused. Water reuse, especially during the dry months, for irrigation could reduce water scarcity.

6.2 Reduction of the Diffuse and Point Pollution Sources

Regarding diffuse pollution, mainly derived from agriculture, the following measures are proposed:

1. Fertilisers should be used strictly in quantities necessary to enrich the soil and to provide it with missing elements. Currently, the overuse of fertilisers has the opposite effects than the ones expected by farmers, both regarding soil fertility and economic yield.
2. Regular crop rotation is significant to maintain the structure and integrity of the soil and improve its productivity.

3. Organic farming is proposed, while Integrated Farming offers a complete farm policy and systems approach to farm management. It seeks to provide efficient and profitable production, which is economically viable and environmentally responsible and delivers safe, wholesome and high-quality food through the efficient management of livestock, forage, fresh produce and arable crops while conserving and enhancing the environment. It is geared towards the optimal and sustainable use of all farm resources such as farm, livestock, soil, water, air, machinery, landscape and wildlife. This is achieved through the integration of natural regulatory processes, on-farm alternatives and management skills, to make the maximum replacement of off-farm inputs, maintain species and landscape diversity, minimise losses and pollution, provide a safe and wholesome food supply and sustain income [102].
4. Drainage canals management. Common reeds (*Phragmites australis* or *Arundo donax*) and in general the vegetation growing in drainage ditches, if managed appropriately, can reduce pollution from agricultural fields.

The most problematic point pollution sources are the wastewaters from olive mills, orange juice production units and from the wastewater treatment plant of Sparta. If these wastewaters are treated appropriately, organic pollution and eutrophication will be significantly reduced. The most appropriate and effective measure will be the prohibition of direct wastewater discharge into the receiving watercourses. Nevertheless, alternative management methods have been proposed, and some are already applied: (1) OMW disposal in evaporation ponds/tanks and use during summer for the irrigation of corn fields and for compost production, (2) by underground disposal of OMW and phytoremediation with poplar trees and (3) in the orange juice production unit, by installing an electrocoagulation unit for the improvement of the wastewater effluent.

6.3 Floods/Droughts Prevention Measures

Significant flood and drought events have occurred historically in the basin, and, therefore, the Prefecture of Laconia has already prepared a Management (Master) Plan [25] for flood protection of the area. The plan has delineated and prioritised the flood-prone areas and suggested a number of sustainable measures that take under consideration mitigation measures for droughts. Some measures to promote restoration of the active river channel and the riparian area could be planned in order to provide synergies between flood protection and biodiversity conservation (e.g. in the river delta).

6.4 *Biodiversity Conservation and Restoration of the Aquatic Ecosystem*

The protection of the basin's biodiversity should be of high priority within a sustainable scheme and should include specific measures:

1. Protection and restoration of the riparian vegetation – The riparian buffer has been proven to prevent the incoming pollution from entering the watercourse [101], serving as a filter, which reduces the incoming pollution load. Maintaining the integrity of the riparian vegetation is of critical importance to the detoxification of the river water in order to upgrade the ecological quality to acceptable levels, especially downstream of the city of Sparta and around the river delta, which were found to be more degraded than the upstream parts of the river.
2. Protection of the river's active channel – Excavators often enter the riverbed to construct flood protection barriers. However, this practice constitutes a recurrent pressure to the aquatic ecosystem, by eliminating fish spawning areas, destroying benthic fauna habitats and removing aquatic vegetation.
3. Establishment of ecological flows downstream of each water pumping and drilling – Water abstracted from springs for irrigation and consumption is led into the water supply network, and thus no water is left for the ecosystem, resulting in the previously described consequences. Public authorities, with the appropriate scientific assistance, should define and provide ecological flows to ensure water flowing in the river throughout the year, as was the situation a few decades ago.
4. Expansion of the protected Natura 2000 area to include an extended zone beyond the river delta, in order to protect the watercourse. The delineation of this protected area exemption must depend on revised biodiversity data and not solely on isolated trigger species; a baseline biodiversity study for the basin needs to be compiled to support conservation measures and decision-making.

The main proposed measures are summarised in Table 2. For each axis, a detailed description of the measures has been carried out to achieve good water quality. Some of the proposed measures, such as the biological farming system, have already been implemented in the ERB. During the LIFE-EnviFriendly project, several technologies for the minimisation of point and non-point sources were demonstrated. In particular, (a) in 'Tzinakos olive mill'; the wastewater is stored in evaporation ponds and used during summer for the irrigation of corn fields and for compost production; (b) in 'Kokkolis olive mill', the wastewater is disposed underground, and phytoremediation with poplar trees is applied; and (c) in the orange juice processing factory, an electrocoagulation unit has been installed for the treatment and improvement of the wastewater. The management of drainage canals as a low-cost agro-environmental measure was also demonstrated. Drainage canals are areas of accumulation of organic debris due to erosion and growth of plants, such as the common reed *Phragmites australis*. The appropriate timing of

Table 2 Main environmental measures proposed in Evrotas River Basin

Axis 1	Modify farming system	Implemented
Axis 2	Alternative choices for water supply	Inter-municipalities companies of drinking water supply[a]
		Wise cost estimate[a]
Axis 3	Drip irrigation and drainage system	Estimation of the real irrigation needs, switching irrigation methods
		Change charges for water abstraction[a]
		Water reuse (municipal and industrial treated wastewater)[a]
Axis 4	Fertiliser control and reduction	Phytoremediation[b]
		Drainage canals management[b]
		Vegetation management on river banks[a]
		Use of fertiliser recommendation system[c]
Axis 5	Estimation zones vulnerable to flooding	Riparian zone stabilisation[b]
		Measures for fire disaster prevention[c]
		Natural hazards procasting[c]
		Management plans for drought and flood protection[c]
Axis 6	Riparian forest protection	River bed protection, remediation/protection of flooded areas[b]
		Ecological effective discharge quantification (during dry period)[a]
		Extension of protection areas to ensure the integrity of biodiversity cores[a]

[a]Under discussion
[b]Active
[c]Has been studied and actions are on the way

cutting may minimise pollutant uptake by plants. Based on the management scenario simulations presented during the LIFE-EnviFriendly project, significant reduction in water abstractions can be achieved without affecting agricultural production. Direct abstraction from the river should be banned. In addition, wastewater treatment should be established at all point sources.

Acknowledgements This chapter is the outcome of a 10-year ecological monitoring survey of the Evrotas River Basin. The results used in this chapter are derived from the European Union projects LIFE-Environment (Environmental Friendly Technologies for Rural Development) [LIFE05 ENV/GR/000245]; MIRAGE (Mediterranean Intermittent River ManAGEment), which received funding from the European Community’s 7th Framework Programme (FP7/2007–2011) under grant agreement No. 211732; and GLOBAQUA (managing the effects of multiple stressors on aquatic ecosystems under water scarcity), which has received funding from the European Union’s 7th Framework Programme for research, technological development and demonstration under grant agreement No. 603629.

References

1. Bory de Saint-Vincent JB (1836) Expédition scientifique de Morée-Section de sciences physiques, vol 1. Levrault, Paris
2. Barnes CLH (2001) Paul Cartledge, Spartan reflections. Bryn Mawr Class Rev. http://bmcr.brynmawr.edu/2002/2002-02-13.html. Accessed 17 Feb 2017
3. Cavanagh W, Crouwel J, Catling RWV, Shipley G (2002) Continuity and change in a Greek rural landscape: the Laconia survey-methodology and interpretation, vol 1. Annual of the British School at Athens, London
4. Mee CG, Cavanagh WG (1998) Diversity in a Greek landscape: the Laconia survey and rural sites projects. In: Cavanagh WG, Walker SEC (eds) Sparta in Laconia. Proceedings of the 19th British museum classical colloquium: B.S.A. studies 4. London, pp 141–148
5. Rackham O (2002) Observations on the historical ecology of Laconia. In: Cavanagh W (ed) Continuity and change in a Greek rural landscape "The Laconia Survey", vol 1. Methodology and interpretation. The Council, British School of Athens, Athens, pp 73–119
6. Phitos D, Constantinidis TH, Kamari G (2009) The red data book of rare and threatened plants of Greece, vols 1, 2. Hellenic Botanical Society, Patras
7. Polunin O (1980) Flowers of Greece and the Balkans, a field guide. Oxford University Press, Oxford
8. Kakouros P (2000) Special environmental study of the Evrotas Delta. Baseline study within the framework of LIFE-nature "Implementation of Management Plans for Pylos Lagoon and Evrotas Delta, Natura 2000 Sites, Greece". Hellenic Ornithological Society, Hellenic Ministry of Environment Land Planning and Public Works, Athens
9. Bizas C (1995) Restoration and management of European riparian landscapes: the case of Evrotas' riparian area. Master's thesis, The European Association for Environmental Management, p 100
10. Dimopoulos P, Bergmeier E, Theodoropoulos K, Eleftheriadou E (2005) Thermophilous deciduous forests in Greece – a preliminary survey. Bot Chron 18:83–100
11. Zogaris S, Tachos V, Chatzinikolaou Y, Bobori D, Koutrakis E, Leonardos I, Anagnopoulos N, Loukatos A, Bourdaniotis N, Kalogianni E, Economou AN (2015) Proposals for possible new sites or site extensions of the Natura 2000 network. Study Section C7, Study 6: "Surveillance and assessment of the conservation status of the ichthyofauna of Community Interest in Greece". Consortium of N. Anagnopoulos/APC, EPEM SA, Ilias Konstantinidis. Unpublished Report. Hellenic Ministry of Environment and Energy, Athens
12. Trigas P, Tsiftsis S, Tsiripidis I, Iatrou G (2012) Distribution patterns and conservation perspectives of the endemic flora of Peloponnese (Greece). Folia Geobot 47(4):421–439
13. Nikolaidis N, Skoulikidis N, Kalogerakis N, Tsakiris K (2009) Environmental Friendly Technologies for Rural Development. Final Report 2005–09, LIFE-ENVIRONMENT LIFE05ENV/Gr/000245 EE (EnviFriendly)
14. Skoulikidis N (2016) The state and origin of river water composition in Greece. In: Skoulikidis N et al. (eds) The rivers of Greece: evolution, current status and perspectives. Springer, Berlin
15. Tzoraki O, Papadoulakis V, Christodoulou A, Vozinaki E, Karalemas N, Gamvroudis C, Nikolaidis NP (2011) Hydrologic modelling of a complex hydrogeologic basin: Evrotas River Basin. In: Lambrakis N et al. (eds) Advances in the research of aquatic environment. Springer, Berlin, pp. 179–186
16. Tselepedakis IG, Theoharatos GA (1989) A bioclimatic classification of the Greek area. Theor Appl Climatol 40:147–153
17. Wagstaff M (2002) The formation of the modern landscape of the survey area, continuity and change in a Greek rural landscape "The Laconia Survey". In: Cavanagh WG, Walker SEC (eds) Methodology and interpretation, vol 1. The Council, British School of Athens, London, pp. 403–420

18. Gamvroudis C (2016) Integrated framework for modeling of hydrology, water quality and sediment transport in river basins with intermittent flow. Department of Environmental Engineering, Technical University of Crete, Chania, 225 pp
19. UNESCO (1979) Map of the world distribution of arid regions. Map at scale 1:25,000,000 with explanatory note. UNESCO, Paris
20. Vagiakakos DB (1989) Evrotas, terminology. In: Proceedings of the special literary symposium, Sparta-Mystras, 27–29 May 1989, pp 361–414
21. Skoulikidis N, Vardakas L, Karaouzas I, Economou AN, Dimitriou E, Zogaris S (2011) Assessing water stress in Mediterranean lotic systems: insights from an artificially intermittent river in Greece. Aquat Sci 73:581–597
22. Zouridakis N, Arnidi E, Stefouli M, Sampatakakis P (2008) Isotopic and hydrogeological study of the groundwater resources of the Evrotas River Basin. DEMO 2008/8 G. NCSR Demokritos, Athens, p 19 (in Greek)
23. Therianos AD (1974) The geographic distribution of river water supply in Hellas. Bull Hell Geol Soc 11:28–58. (in Greek)
24. Skoulikidis N (2009) The environmental state of rivers in the Balkans: a review within the DPSIR framework. Sci Total Environ 407:2501–2516
25. Mariolakos I, Andreadakis E, Sambaziotis E, Kapourani E, Karagkiozi E (2007) Strategic planning of anti-flooding protection of Lakonia Prefecture. Technical Report. National and Kapodistrian University of Athens, Lakonia Prefecture, 93 pp (in Greek)
26. Dafis S, Papastergiadou E, Georghiou K, Babalonas D, Georgiadis T, Papageorgiou M, Lazaridou T, Tsiaoussi V (1996) The Greek 'habitat' project NATURA 2000: an overview. Life Contract B4-32000/94/756, Commission of the European Communities. The Goulandris Natural History Museum and Greek Biotope/Wetland Centre, Kifisia and Thessaloniki
27. Sfikas G (2001) Botanic heavens of Greece. Toubi's SA, Athens, 256 pp (in Greek)
28. Skoulikidis N, Economou A, Karaouzas I, Vardakas L, Gritzalis K, Zogaris S, Dimitriou E, Tachos V (2008) Hydrological and biogeochemical monitoring in Evrotas River Basin final technical report, HCMR. In: Nikolaidis et al (eds) Environmental Friendly Technologies for Rural Development. Project ENVI-FRIENDLY LIFE-ENVIRONMENT: LIFE 05 ENV/GR/000245, Environmental Friendly Technologies for Rural Development
29. Economou AN, Giakoumi S, Vardakas L, Barbieri R, Stoumboudi M, Zogaris S (2007) The freshwater ichthyofauna of Greece: an update based on a hydrographic basin survey. Mediterr Mar Sci 8:91–168
30. Zogaris S, Economou AN, Dimopoulos P (2009) Ecoregions in the Southern Balkans: should they be revised? Environ Manag 43:682–697. doi:10.1007/s00267-008-9243-y
31. Koumpli-Sovantzi L (1997) The Bryophyte flora of Greece. Flora Mediterr 7:173–179
32. Korakis G, Gerasimidis AC (2006) Coastal and halophytic habitats and their flora in Evrotas Delta (SE Peloponnisos, Greece). Biol Res 6:155–166
33. Gloer P, Fralnioski A, Pesic V (2010) The Bithyniidae of Greece (Gasteropoda: Bithyniidae). J Conchol 40:179–187
34. Gritzalis KC (2009) First record of the species *Melanopsis praemorsa* (LINNE 1758), Mollusca; Gastropoda; Sorbeoconcha, in the Greek mainland (Vasilopotamos R., Laconia, Peloponnese, Greece). In: Proceedings of the 31st conference of the Hellenic Society for Biological Sciences, pp 62–63
35. Karaouzas I (2011) Ecotoxicity and effects of agro-industrial wastewaters to stream biodiversity at multiple levels of biological organization. PhD thesis, University of Ioannina
36. Karaouzas I, Skoulikidis N, Giannakou O, Albanis TA (2011) Spatial and temporal effects of olive mill wastewaters to stream macroinvertebrates and aquatic ecosystems status. Water Res 45:6334–6346
37. Karaouzas I (2009) The larva of *Hydropsyche peristerica* Botosaneanu & Marinković-Gospodnetić 1968, (Trichoptera, Hydropsychidae) with some notes on its ecology. Aquat Insects 31(2):139–143. doi:10.1080/01650420802623786

38. Karaouzas I (2014) The larva of *Rhyacophila loxias* Schmid, 1970 (Trichoptera: Rhyacophilidae) with notes on its ecology. Aquat Insects 36(1):15–21. doi:10.1080/01650424.2014.999793
39. Hupało K, Mamos T, Olszewska W, Wysocka A, Karaouzas I, Plaiti W, Michał Grabowski M (2015) Diversity and origin of freshwater gammarids from crete and peloponnese – preliminary results. In: Poulakakis N et al (eds) 2015. Abstracts of the International Congress on the Zoogeography and Ecology of Greece and Adjacent Regions, 13th ICZEGAR, 7–11 Oct 2015, Irakleio, Crete, Greece. Hellenic Zoological Society, 203 pp
40. Karaouzas I, Andriopoulou A, Kouvarda T, Murányi D (2016) An annotated checklist of the Greek stonefly fauna (Insecta: Plecoptera). Zootaxa 4111(4):301–333. doi:10.11646/zootaxa.4111.4.1
41. Karaouzas I, Płóciennik M (2016) Spatial scale effects on Chironomidae diversity and distribution in a Mediterranean River Basin. Hydrobiologia 767:81–93. doi:10.1007/s10750-015-2479-7
42. Bianco P (1990) Potential role of the paleohistory of the Mediterranean and Paratethys basins on the early dispersal of Euro-Mediterranean freshwater fishes. Ichthyiol Explor Freshwater 1:167–184
43. Economidis PS, Banarescu PM (1991) The distribution and origins of freshwater fishes in the Balkan Peninsula, especially in Greece. Int Rev Gesamten Hydrobiol Hydrogr 76:257–283
44. Oikonomou A, Leprieur F, Leonardos ID (2014) Biogeography of freshwater fishes of the Balkan Peninsula. Hydrobiologia 738(1):205–220
45. Barbieri R, Zogaris S, Kalogianni E, Stoumboudi MTH, Chatzinikolaou Y, Giakoumi S, Kapakos Y, Kommatas D, Koutsikos N, Tachos V, Vardakas L, Economou AN (2015) Freshwater fishes and lampreys of Greece: an annotated checklist. Monographs on marine sciences, no. 8. Hellenic Centre for Marine Research, Athens
46. IUCN (2016) The IUCN red list of threatened species. Version 2016-3. http://www.iucnredlist.org. Accessed 17 Feb 2017
47. Stephanidis A (1971) On some freshwater fish of Greece. Biol Gallo-Hell 3(2):213–241
48. Kottelat M, Barbieri R (2004) *Pseudophoxinus laconicus*, a new species of minnow from Peloponnese, Greece, with comments on the West Balkan *Pseudophoxinus* species (Teleostei: Cyprinidae). Ichthyol Explor Freshwater 15(2):147–160
49. Tsigenopoulos CS, Karakousis Y (1996) Phylogenetic relationships of *Leuciscus keadicus*, an endemic cyprinid species from Greece, with other species of the genus *Leuciscus*. Folia Zool 45:87–93
50. Doadrio I, Carmona JA (1998) Genetic divergence in Greek populations of the genus Leuciscus and its evolutionary and biogeographical implications. J Fish Biol 53:591–613
51. Zardoya R, Economidis PS, Doadrio I (1999) Phylogenetic relationships of Greek Cyprinidae: molecular evidence for at least two independent origins of the Greek cyprinid fauna. Mol Phylogenet Evol 13:122–131
52. Perea S, Böhme M, Zupančič P, Freyhof J, Šanda R, Özuluğ M, Abdoli A, Doadrio I (2010) Phylogenetic relationships and biogeographical patterns in Circum-Mediterranean subfamily Leuciscinae (Teleostei, Cyprinidae) inferred from both mitochondrial and nuclear data. Evol Biol 10:265
53. Durand JD, Ûnlï E, Doadrio I, Pipoyan S, Templeton AR (2000) Origin, radiation, dispersion and allopatric hybridization in the chub *Leuciscus cephalus*. Proc R Soc Lond Ser B 267:1687–1697
54. Kottelat M, Freyhof J (2007) Handbook of European freshwater fish. Kottelat, Cornol and Freyhof, Berlin
55. Barbieri R, Economou AN, Stoumboudi MTH, Economidis PS (2002) Freshwater fishes of Peloponnese (Greece): distribution, ecology and threats. In: Collares-Pereira M et al. (eds) Conservation of freshwater fishes: options for the future. Fishing News Books, Blackwell Science, Oxford

56. Vardakas L, Kalogianni E, Papadaki C, Vavalidis T, Mentzafou A, Koutsoubas D, Skoulikidis N (2017) Defining critical habitat conditions for the conservation of three endemic and endangered cyprinids in a Mediterranean intermittent river prior to the onset of drought. Aquat Conserv. doi:10.1002/aqc.2735
57. Pires AM, Cowx IG, Coelho MM (2000) Life history strategy of *Leuciscus pyrenaicus* (Cyprinidae) in intermittent streams of the Guadiana basin (Portugal). Cybium 24:287–297
58. Santos JM, Ferreira MT (2008) Microhabitat use by endangered Iberian cyprinids *Iberochondrostoma almacai* and chub *Squalius aradensis*. Aquat Sci 70:272–281
59. Economou AN, Barbieri R, Stoumboudi M (1999) Threatened endemic freshwater fishes to Greece: the Evrotas case. Workshop on ‘Wetland Restoration’, 12–14 Nov 1999, Gythion, Greece
60. Vardakas L, Kalogianni E, Zogaris S, Koutsikos N, Vavalidis T, Koutsoubas D, Skoulikidis NT (2015) Distribution patterns of fish assemblages in an Eastern Mediterranean intermittent river. Knowl Manag Aquat Ecosyst 416:30. doi:10.1051/kmae/2015026
61. Valakos ED, Pafilis P, Sotiropoulos K, Lymberakis P, Maragou P, Foufopoulos J (2008) The amphibians and reptiles of Greece. Frankfurt contributions to natural history, vol 32. Ed Chimaira, Frankfurt, p. 563
62. Portolou D, Bourdakis S, Vlachos C, Kastritis T, Dimalexis T (2009) The important bird areas of Greece: priority areas for biodiversity conservation. Hellenic Ornithological Society, Athens. (in Greek)
63. Legakis A, Maragou P (2009) The red data book of threatened animals of Greece. Hellenic Zoological Society, Athens, p. 528. (in Greek)
64. Giannatos G, Marinos Y, Maragou P, Catsadorakis G (2005) The status of the golden jackal (*Canis aureus* L.) in Greece. Belg J Zool 135:145–149
65. Galanaki A (2000) A study of the Otter (*Lutra lutra* L.) in the Evrotas River Delta, Greece. MSc thesis, University of Wales, Bangor
66. Dimopoulos P, Zotos A, Zogaris S (2009) Protocol for a rapid assessment of riparian zone conservation. Environmental Friendly Technologies for Rural Development, Final Report 2005–09, LIFE-ENVIRONMENT LIFE05ENV/Gr/000245 EE (EnviFriendly), pp 127–141
67. Angelidis MO, Markantonatos PG, Bacalis NC (1995) Impact of human activities on the quality of river water: the case of Evrotas River catchment basin, Greece. Environ Monit Assess 35:137–153
68. Karaouzas I, Lambropoulou DA, Skoulikidis N, Albanis TA (2011) Levels, sources and spatiotemporal variation of nutrients and micropollutants in small streams of a Mediterranean River Basin. J Environ Monit 13:3064–3074. doi:10.1039/C1EM10637F
69. Valta K (2008) Restoration of pollutants and assessment of the chemical quality of the Evrotas river basin. Master thesis, Technical University of Crete, Chania
70. Karaouzas I (2017) Agro-industrial wastewater pollution in Greek river ecosystems. In: Skoulikidis N et al. (eds) The rivers of Greece: evolution, current status and perspectives. Springer, Berlin
71. Karaouzas I, Cotou E, Albanis TA, Kamarianos A, Skoulikidis NT, Giannakou U (2011) Bioassays and biochemical biomarkers for assessing olive mill and citrus processing wastewater toxicity. Environ Toxicol 26:669–676. doi:10.1002/tox.20606
72. Rouvalis A, Theodoropoulos C, Iliopoulou-Georgudaki J (2013) Assessment of toxicity of the untreated and Pleurotus ostreatus treated olive mill wastewater by using microbiotests. Int J Environ Eng 5:373–386
73. Vlyssides AG, Loizidore M, Gimouhopoulos U, Zorpas A (1998) Olive oil processing wastes production and their characteristics in relation to olive oil extraction methods. Fresenius Environ Bull 7:308–313
74. Blake W, Theocharopoulos S, Skoulikidis N, Clark P, Tountas P (2008) Wildfire, soil erosion and the risk to aquatic resources: evidence from the burnt Evrotas River basin, southern Peloponnese, Greece. In: Geophysical research abstracts

75. Karaouzas I, Skoulikidis NT, Blake W (2012) Short-term effects of wildfires on stream benthic communities in an intermittent Mediterranean stream. In: Legakis A et al (eds) Abstracts of the international congress on the zoogeography and evolution of Southeastern Europe and the Eastern Mediterranean, 12th ICZEGAR, 18–22 June 2012, Athens, Greece. Hellenic Zoological Society, 230 pp
76. Skoulikidis N, Amaxidis Y, Bertahas I, Laschou S, Gritzalis K (2006) Analysis of factors driving stream water composition and synthesis of management tools – a case study on small/ medium Greek catchments. Sci Total Environ 362:205–241
77. Lampou A, Skoulikidis N, Papadoulakis V, Vardakas L (2015) Evaluation of the waters' condition in the final receiver of the wastewater treatment plant in Municipality of Sparta. In: Proceedings of the 11th Panhellenic conference of oceanography & fisheries, pp 661–664
78. Skoulikidis N, Amaxidis Y (2009) Origin and dynamics of dissolved and particulate nutrients in a minimally disturbed Mediterranean river with intermittent flow. J Hydrol 37:218–229
79. EEA (2012) European waters – assessment of status and pressures. EEA Report No 8/2012. European Environment Agency, Copenhagen
80. Skoulikidis NT, Sabater S, Datry T, Morais M, Buffagni A, Dörflinger G, Zogaris S, Sánchez-Montoya MM, Bonada N, Kalogianni E, Rosado J, Vardakas L, De Girolamo AM, Tockner K (2017) Non-perennial Mediterranean rivers in Europe: status, pressures, and challenges for research and management. Sci Total Environ 577:1–18. doi:10.1016/j.scitotenv.2016.10.147
81. Baldwin DS, Mitchell AM (2000) The effects of drying and re-flooding on the sediment and soil nutrient dynamics of lowland river-floodplain systems: a synthesis. River Res Appl 16:457–467
82. Hu BL, Shen LD, Xu XY, Zheng P (2011) Anaerobic ammonium oxidation (anammox) in different natural ecosystems. Biochem Soc Trans 39:1811–1816
83. Nimick DA, Gammons CH, Parker SR (2011) Diel biogeochemical processes and their effect on the aqueous chemistry of streams: a review. Chem Geol 283:3–17
84. Skoulikidis NTH, Vardakas L, Amaxidis Y, Michalopoulos P (2017) Biogeochemical processes controlling aquatic quality during drying and rewetting events in a Mediterranean non-perennial river reach. Sci Total Environ 575:378–389
85. Amalfitano S, Fazi S, Zoppini A, Barra Caracciolo A, Grenni P, Puddu A (2008) Responses of benthic bacteria to experimental drying in sediments from Mediterranean temporary rivers. Microb Ecol 55:270–279
86. Corti R, Datry T, Drummond L, Larned ST (2011) Natural variation in immersion and emersion affects breakdown and invertebrate colonization of leaf litter in a temporary river. Aquat Sci 73:537–550
87. Foulquier A, Artigas J, Pesce S, Datry T (2015) Drying responses of microbial litter decomposition and associated fungal and bacterial communities are not affected by emersion frequency. Freshwater Sci 34:1233–1244
88. McClain ME, Boyer EW, Dent CL, Gergel SE, Grimm NB, Groffman PM, Hart SC, Harvey JW, Johnston CA, Mayorga E, McDowell WH, Pinay G (2003) Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems. Ecosystems 6:301–312
89. McDonough OT, Hosen JD, Palmer MA (2011) Temporary streams: the hydrology, geography and ecology of non-perennial flowing waters. In: Eliot HS, Martin LE (eds) River ecosystems: dynamics, management and conservation. Nova Science, New York
90. Mummey DL, Smith JL, Bolton Jr H (1994) Nitrous oxide flux from a shrub-steppe ecosystem: sources and regulation. Soil Biol Biochem 26:279–286
91. Peterjohn WT, Schlessinger WH (1991) Factors controlling denitrification in a Chihuahuan desert ecosystem. Soil Sci Soc Am J 55:1694–1701
92. European Commission (2013) Directive 2013/39/EU of the European parliament and of the council of 12 August 2013 amending directives 2000/60/EC and 2008/105/EC as regards priority substances in the field of water policy. Off J Eur Union L226:1–17

93. WFD CIS (2003) Common implementation strategy for the Water Framework Directive (2000/60/EC). Guidance document No. 13: overall approach to the classification of ecological status and ecological potential
94. Environment Agency (2003) River habitat survey in Britain and Ireland. Field survey guidance manual. Environmental Agency, Bristol
95. Artemiadou V, Lazaridou M (2005) Evaluation score and interpretation index for the ecological quality of running waters in central and northern Hellas. Environ Monit Assess 110:1–40
96. Haury J, Peltre MC, Trémolières M, Barbe J, Thiébaut G, Bernez I, Daniel H, Chatenet P, Haan-Archipo G, Muller S, Dutartre A, Laplace-Treyture C, Cazaubon A, Lambert-Servien E (2006) A new method to assess water trophy and organic pollution – the macrophyte biological index for rivers (IBMR): its application to different types of river and pollution. Hydrobiologia 570:153–158
97. Lazaridou M, Ntislidou C, Karaouzas I, Skoulikidis N (2017) Establishment of a national biotic index for the ecological status assessment of rivers in Greece. In: 10th symposium for European freshwater sciences, Olomouc, Czech Republic, 2–7 July. Book of abstracts
98. Prygiel J, Coste M (2000) Guide méthodologique pour la mise en oeuvre de l'Indice Biologique Diatomées. NF T 90–354. Agences de l'eau – Cemagref, Bordeaux, France, 133 pp
99. Tachos V, Zogaris S, Koutsikos N, Vardakas L, Kommatas D, Chatzinikolaou Y, Kalogianni E, Kalaitzakis N, Economou AN, Schmutz S (2016) Developing a national fish index for the assessment of the ecological quality of lotic waters in Greece: elaboration of a multi-metric model. Proc Hell Confer Ichthyol 16:333–336
100. Gamvroudis C, Nikolaidis NP, Tzoraki O, Papadoulakis V, Karalemas N (2015) Water and sediment transport modeling of a large temporary river basin in Greece. Sci Total Environ 508:354–365
101. Theodoropoulos C, Iliopoulou-Georgudaki J (2010) Response of biota to pollution and water quality degradation along two medium-sized river basins in southwestern Greece. Ecol Indic 10:1231–1238
102. European Initiative for Sustainable Development in Agriculture (EISA) (2012) European Integrated Farming Framework. A European definition and characterisation of Integrated Farming (IF) as guideline for sustainable development of agriculture. http://www.sustainable-agriculture.org/wp-content/uploads/2012/08/EISA_Framework_english_new_wheel_170212.pdf. Accessed 17 Feb 2017
103. Pausanias. Pausanias description of Greece (Translated by Jones WHS, Omerod HA. Loeb classical library volumes. Harvard University Press, Cambridge; William Heinemann Ltd., London, 1918)

Macroinvertebrate Assemblages and Biological Status of Rivers in Northern and Central Greece

Lazaridou Maria and Ntislidou Chrysoula

Abstract This paper investigates the benthic macroinvertebrate fauna of the northern and central rivers of Greece and their use in the assessment of the biological/ecological conditions of water bodies towards the fulfilment of the Water Framework Directive for good ecological status/potential by the end of 2015. The macrozoobenthos from reference or moderately disturbed sites did not significantly differ as to their richness, diversity and sensitivity among sites. Altitude, among other environmental parameters, was the differentiating parameter according to the canonical correspondence analysis (CCA) of the structure/composition of benthic macroinvertebrates. Seasonality exists in high- and low-altitude reference sites for sensitive to organic pollution taxa. The results of the STAR_ICMi and HESY indices coincided totally when the ecological quality was below good. More than 70% of the sites were characterised as of lower than good ecological status/potential. According to HESY, water quality varied according to the altitude from upstream to downstream sites (pollution gradient). The application of operational monitoring or continuous programme of measures is needed for most of the basins in order to meet the environmental objectives and the risk management (IMPRESS analysis) according to the WFD. Finally, the DPSIR framework shows that the 'drivers' agriculture, livestock and sewage untreated effluents cause the deterioration of the ecological quality of water and habitat degradation ('state').

Keywords DPSIR, Greece, IMPRESS, Macroinvertebrates, River subbasin/basin

---

L. Maria (✉) and N. Chrysoula
Laboratory of Zoology, Department of Biology, Faculty of Sciences, Aristotle University of Thessaloniki, 54124 Thessaloniki, Greece
e-mail: mlazarid@bio.auth.gr; chntisli@windowslive.com

N. Skoulikidis et al. (eds.), *The Rivers of Greece: Evolution, Current Status and Perspectives*, Hdb Env Chem (2018) 59: 327–354, DOI 10.1007/698_2015_445,

Contents

1 Introduction

In a few months from now, member states are supposed to have achieved at least a 'good ecological status' for natural bodies and 'good ecological potential', for heavily modified and artificial water bodies according to the Water Framework Directive (WFD, 2000/60/EC). The inclusion of explicit ecological reports for every target and deliverable makes the WFD unique in its implementation and delivery [1].

The originality of this WFD is the use of hydromorphological and physico-chemical quality elements which have to support the biological ones, for the ecological quality assessment of surface waters. In order for member states to come up with comparable results, they must present the status/potential in a colour class system: 'high' (blue only for the status), 'good' (green), 'moderate' (yellow), 'poor' (orange) and 'bad' (red). For this assessment, the observed quality elements must be compared to the reference (undisturbed) ones [ecological quality ratio (EQR)]. For the above ratio, surface waters must be classified into types following the System A or B. Type-specific reference conditions are the basis of the classification schemes and subsequent issues for the implementation of the WFD [2]. One of the biological elements used for the quality assessment is the study of the composition, abundance and proportion of sensitive to tolerant taxa of benthic macroinvertebrates. Based on this, several scores and indices have been created to assess the ecological quality (BMWP [3], IBMWP [4], STAR_ICMi [5], Hellenic Evaluation System [6]).

Additionally, pressures and impacts play a key role in the likelihood that a water body will or will not meet the set objectives of the WFD. The analyses of pressures and impacts are crucial to be developed if appropriate programmes of measures have to be designed and implemented [7]. The driver-pressure-state-impact-response (DPSIR) approach was established as a possible analytical framework for determining pressures and impacts under the WFD [7–9] and identifying the cause-effect relationships between the environment and various anthropogenic activities in a wider socio-economic context [10]. The WFD also introduced the requirement to evaluate new methodological approaches for the development of strategies contributing to the sustainable water resources management [8].

This paper concerns sites coming from northern and central Greek rivers sampled during the last 21 years. Comparisons are made (a) to reference samples as to altitude, seasonality and sensitivity; (b) among sites of the same type as to their ecological quality, status/potential, the morphological modifications and the likelihood of these basins not to meet the objectives of the WFD according to an IMPRESS analysis up to 2015; and (c) the results of the DPSIR framework among the different subbasins/basins.

2 Methods and Materials

2.1 Databases

Four databases have been constructed from studies conducted in subbasins/basins of northern and central Greece during the past 21 years, which include information concerning the following: (a) benthic macroinvertebrates, (b) in situ physico-chemical measurements and/or analyses in the laboratory, (c) in situ hydro-morphological surveyed elements (in this database, the likelihood of meeting the objectives of the WFD according to IMPRESS analysis was also included in each basin/subbasin) and (d) the data of type-specific conditions only from reference sites. The above studies were performed by members/students of the Laboratory of Zoology, School of Biology, and of the Interdisciplinary Postgraduate Programme of Aristotle University of Thessaloniki entitled: "Ecological Quality and Management at a basin level" of the Schools of Biology, Geology and Civil Engineering (Table 1).

2.2 Sampling Methods

The benthic macroinvertebrates were sampled using a 250×230 mm, D-shaped pond net (0.9-mm mesh size [60]) according to the semi-quantitative 3-min kick/sweep method [61] plus a 1-min search applied when bank vegetation existed [62, 63]. During the 3 min, all the microhabitats were covered proportionally according to the matrix of 64 possible different river habitats which could be

Table 1 Sources of literature data used for the purpose of the databases

River	Number of sites	Number of samples	References
Aggitis	9	48	[11–14]
Aliakmonas	60	112	[12–19]
Aoos/Vjose	17	33	[20]
Axios	46	113	[19, 21–27]
Chavrias	9	15	[28]
Dadia	18	26	[29, 30]
Kosynthos	9	9	[31]
Mavroneri	10	10	[32]
Olynthios	7	7	[33]
Nestos	11	23	[34, 35]
Pineios	77	77	[36]
Pieria	7	7	[37]
Pineios streams	11	21	[38, 39]
Skouries and Olympiada streams	25	92	[40, 41]
Samothraki	6	10	[42]
Sofaditis	17	28	[43, 44]
Strymonas	21	41	[45–48]
Streams of Mygdonia Basin	21	21	[49–52]
Streams of Plastira Basin	11	76	[53–55]
Travos	3	3	[56]
Tripotamos	10	20	[57, 58]
Ziliana	4	4	[59]
Total 25	409	796	50

observed in a site [21]. Specimens were identified mainly down to the family level except for Ostracoda, Hydracarina, Araneae and Oligochaeta (apart from Tubificidae).

Dissolved oxygen (DO mg/l), water temperature (WTemp, °C) pH and conductivity (μS/cm) were measured in situ with appropriate probes. Total suspended solids (TSS, mg/l), nutrients ($N–NO_2$, $N–NH_3$, $N–NH_4$ and $P–PO_4$, mg/l) and biological oxygen demand (BOD_5, mg/l) were estimated following APHA [64]. Flow was measured with a flowmeter (type FP101) and stream discharge (m^3/s) was calculated for each station. The percentage composition of the substrate was visually estimated according to the Wentworth [65] scale. The Habitat Modification Score (HMS) from the River Habitat Survey [66, 67] was calculated to assess the extent of human alterations.

2.3 Typology

In Greece there is no national river typology system for the determination of reference conditions. In these studies, System B according to the WFD 2000/60 EC was followed because the Axios/Vardar river basin (a transboundary Greek-FYROM river) belongs to two different ecoregions (ecoregions 6 and 7) according to System A. To distinguish the water bodies of the studied basins, the obligatory descriptors were selected, and the slope from the optional ones was included. Furthermore, a new category in the basin descriptor was added (0–10 km^2), since a large number of small but substantial rivers exist in Greece. Typology was applied to ensure that sampling sites from upstream and downstream point sources of pollution or tributaries, flowing into the main course, belong to the same type so that a comparison of their quality is possible. The rivers were also characterised according to the Mediterranean intercalibration typology [68, 69] in one of the five types (R-M1, R-M2, R-M3, R-M4 and R-M5). Lately, the final European Commission 2013/480 proposal has excluded the altitude descriptor from all the intercalibration river types.

2.4 Selection of Reference Sites

In order to create the dataset of the northern and central Hellenic reference sites, the overall European criteria of the intercalibration technical report [69], approved and applied over Europe, were adopted. The criteria were applied at a catchment reach and sampling site scale, as described in Ntislidou et al. [70]. For smoother compliance with Mediterranean peculiarities, the criteria proposed by Sánchez-Montoya et al. [71] in Spain were also used. Initially, the choice of the reference sites/samples was based on the absence of non-natural land uses at reach scale and minor hydromorphological modifications at sampling site scale. Consequently, Corine Land Cover 2000 was applied to calculate the percentage cover of land uses upstream each site (e.g. % artificial, % intensive agricultural, % extensive agricultural, % forest and seminatural, % wetlands and % water bodies areas). The chosen samples were then checked for meeting the nutrient criteria proposed by Bonada et al. [72] used in the Spanish programme GUADALMED (HID98-0323-C05 and REN2001-3438-C07) and by Munné et al. [73] in the rivers of Catalonia (Spain). Reference sites were estimated with high/good quality based on benthic macroinvertebrates according to the Hellenic Evaluation System (HESY) [6].

The relationship of macrozoobenthos taxa from reference sites with environmental parameters was performed with the canonical correspondence analysis (CCA, with the CANOCO version 4.5.1 software) [74] because the length of the gradient of the first theoretical parameter (of the first ordination axis) of detrended correspondence analysis (DCA) was three or more times greater than the standard deviation of the benthic macroinvertebrates [75]. The Monte Carlo test ($p < 0.05$)

and the inflation factor (<20) were taken into consideration in the selection of the environmental parameters used in CCA [altitude (m), conductivity (μs/cm), temperature (°C), pH, DO (mg/l), N–NO_2 and N–NH_4 (mg/l)].

Seasonality of benthic macroinvertebrates has been analysed in the reference sites during the low and high flow period. The sensitivity of macrozoobenthos in relation to organic pollution was based on the scores of the HESY (sensitive to pollution, 120–80; medium, 78–50; and tolerant 40–1 score) [6].

The Kruskal-Wallis test (SPSS 20) was applied to the abundance, diversity and sensitivity of the macrozoobenthos from reference sites in order to determine whether there is a significant discrepancy among them.

2.5 Assessment of the Ecological Quality

In order to assess the good ecological status (GES) in natural water bodies and good ecological potential (GEP) in heavily modified water bodies (HMWB), Guidance No. 13 [76] was followed. For the determination of the GES, biological data are examined for deviations from reference conditions, and subsequently the deviation of the physico-chemical and the hydromorphological data are taken into consideration. For the GEP, the hydromorphological potential is firstly checked and then the biological and physico-chemical data. HMWB were not distinguished as separated types but were integrated with natural water bodies having comparable typology descriptors and categories [77]. Both arrive at a five-class quality assessment. In this paper, the GES and GEP have been determined for 796 samples from northern and central Greece. The Hellenic Evaluation System (HESY) was used for the quality assessment [6], jointly with the multimetric STAR_ICMi index [5, 78]. The permitted levels for the support of the fish life [79] were used for the assessment of the physico-chemical values at each site. Finally, in order to meet the high status/potential for the hydromorphological conditions, the HMS values were used, grouped into three categories [HMS, 0–2 (pristine or seminatural), 3–8 (predominantly unmodified, >8 (obviously or significantly or severely modified)].

The HESY is based on a dataset of 143 reference samples from all river types (R-M1, 2, 3, 4, 5) and 330 from less than good quality of seven different river basins. The family identification level is applied to all zoobenthos taxa [except for Ostracoda, Hydracarina, Araneae and Oligochaeta (apart from Tubificidae)]. The HESY is composed of more families than the other European evaluation systems [BMWP [3], IBMWP [4], BBI [80], etc.]. Moreover, it takes into consideration the tolerance, the abundance/richness and the habitat diversity of the biocommunities, all requirements of the WFD. It is composed of (a) the Hellenic Evaluation Score (HES) which is a BMWP-type score [3], (b) the average HES (AHES) which is similar to ASPT [3] and (c) the SemiHES (the semi-total of the HES and AHES values resulting in the final HESY) which is standardised against the habitat diversity richness matrix (GHRM) [21]. The SemiHES is interpreted at a five-class scale according to the WFD [6]. The HESY is intercalibrated for the R-M4 [53] and R-M1 and R-M2 river types [70].

The multimetric index STAR_ICMi expresses the tolerance, the habitat diversity and the diversity/richness of the benthic macroinvertebrates community, describing the gradients effectively and discriminating between different quality classes which can be calculated from a wide range of geographical contexts [81]. The median values for each Intercalibration Common Metrics of Hellenic reference samples were determined by [53, 70]. These values were used to transform the results of biological metrics to the EQR multimetric index STAR_ICMi according to the methodology proposed by the Mediterranean Intercalibration Group [81]. Finally, the quality of each station was determined by the officially set quality boundaries [78].

Linear regression analysis (SPSS 20) was applied between SemiHES results and the distance of each site from the source in order to study the possible longitudinal pollution trend. The latter was applied only on the main course of the rivers with no tributaries in between that may change the water quality and the existence of a trend.

2.6 IMPRESS Analysis, Risk Assessment and DPSIR Framework

IMPRESS analysis is based on pressures and morphological alterations. The domestic wastewater and the livestock wastewater are considered as point sources of pollution. In order to calculate the emissions from the domestic wastewaters, their treatment (secondary or tertiary wastewater treatment) and/or the existence of septic tanks and the total number of human population according to the Hellenic Statistical Authority (ELSTAT, http://www.statistics.gr) from the census of 2001 were taken into consideration. For the livestock waste, the breeding animals from the census of 2001 (ELSTAT, http://www.statistics.gr) were taken into account. The emissions of human population were calculated according to the factors of Fribourg-Blanc and Courbet [82] and Andreadakis et al. [83], whereas the ones from the livestock were estimated according to [84]. The emissions from livestock transported to the surface waters are 20% for BOD, 15% for N and 3% for P [83]. Diffuse sources of pollution were determined using the Corine Land Cover 2000, and their emissions were calculated according to the factors of Andreadakis et al. [83].

The pressures from pollution sources are significant if the total emissions exceed the permitted limits for irrigation (e.g. [85]) and/or for fish life [79], after being adjusted to the mean annual flow of each river basin. The morphological alterations are significant if (a) agricultural land cover is more than 40% of the length of the river course [86], (b) urban land cover is more than 2.5% [87] of the total extent of the river basin and (c) HMS is more than 8. The impact assessment and the evaluation of likelihood of failing to meet the environmental objectives and the risk management were based on the methodology proposed by Castro et al. [88].

The DPSIR is a chain of causal links starting from ‘drivers’ (causes) through ‘pressures’ (e.g. pollutants), ‘state’ (physical, chemical, biological) and ‘impacts’ on ecosystems (structure and function) in order to lead to ‘responses’ (policy) (Guidance document 3 [89]). This framework was firstly used in 1995 by the European Environment Agency and by Eurostat, in order to organise the environmental indicators and statistics [90]. Natural, social and economics sciences are combined together under one analysis for management, and DPSIR considers human activities an essential part of the ecosystem [91]. In this study, the DPSIR framework was performed in 23 basins in order to assess the effects of human activities on ecosystems, trying to relate human drivers like agricultural activities with pressures of nutrients and impact on physico-chemical values and bio-communities and the institutional response in terms of regulatory legislation for protection.

3 Results

3.1 Intercalibration River Types and Reference Sites

According to Corine Land Cover 2000, the reference sites had minimum artificial surfaces and low percentage of nonirrigated or permanently irrigated land (Fig. 1). In Fig. 1, the values of nutrient parameters of reference samples are also represented. The reference sites based on the HESY had high quality (except for 3 out of 49 samples which displayed good quality in R-M2 river type).

The northern and central Hellenic reference database consists of 95 reference samples (from 36 sites) (Fig. 2) from all R-M types (1, 2, 3, 4, 5). Among the above samples, 30 (belonging to 9 sites) were ascribed to the R-M1 river type, 16 samples (from 10 sites) to the R-M2, one site/sample to the R-M3, 34 samples (from 9 sites) to the R-M4 and 14 samples (from 7 sites) to the R-M5 according to the European Decision 2008/915 [92]. Some of the sites/samples did not meet the criteria of the above decision precisely; however, they ecologically belonged to the proposed river type [e.g. in the R-M1 type, samples derived from an altitude lower than 200 m or higher than 800 m (up to 1,000 m); in the R-M2 type some samples were taken from an altitude higher than 600 m (up to 1,000 m)] which is in accordance with the last European Commission decision (2013/480) in which the altitude is excluded as a descriptor from all Mediterranean intercalibration river types.

About 50% of the families overlapped between the different R-M types showing that no clear type-specific macrozoobenthos taxa at a family level exist [93]. Altitude was the most differentiating descriptor, affecting the structure of benthic macroinvertebrates (Table 2) at the level of the reference sites according to the

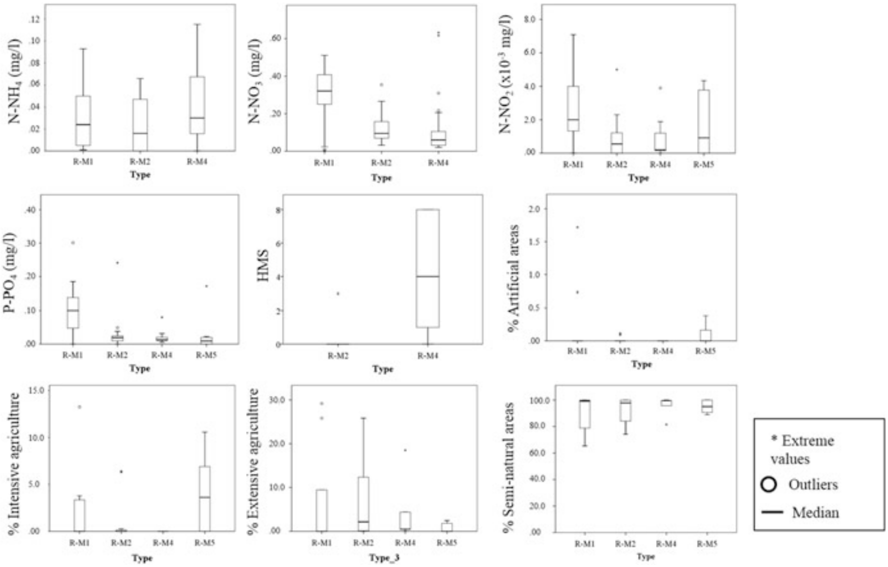


Fig. 1 *Boxplots* representing the distribution of nutrients and HMS values and land use in the reference sites

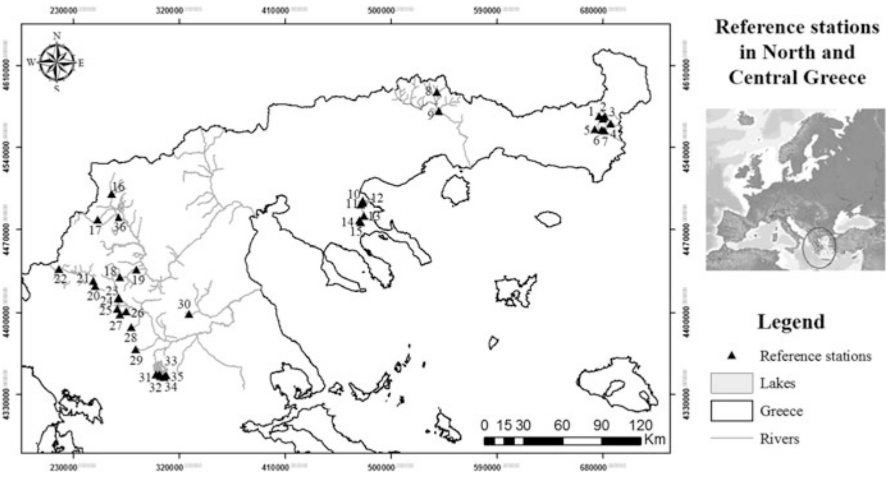


Fig. 2 Reference sites from northern and central Greece

canonical correspondence analysis. The macrozoobenthos from reference or moderately disturbed sites did not significantly differ as to the richness, diversity and sensitivity among sites (Kruskal-Wallis, $p > 0.05$).

Table 2 Interset correlations of environmental variables [statistically significant ($p < 0.05$) according to Monte Carlo Test] from the canonical correspondence analysis

Environmental variables	Axes			
	1	2	3	4
Altitude (m)	**0.878**	0.043	0.205	0.421
Conductivity (μS/cm)	0.303	0.323	0.151	0.054
Temperature (°C)	0.304	**0.748**	0.187	0.443
pH	0.220	0.683	0.504	0.172
DO (mg/l)	0.276	0.064	0.093	**0.457**
$N–NO_2$ (mg/l)	0.390	0.097	0.355	0.024
$N–NH_4$ (mg/l)	0.334	0.440	**0.769**	0.283
Eigenvalues	0.185	0.121	0.094	0.083
Species-environment correlations	0.867	**0.864**	0.674	0.746
Cumulative percentage variance of species data	5.6	9.2	12.0	14.5
Cumulative percentage variance of species-environment relation	31.5	**52.2**	68.2	82.3
Test of significance of first canonical axis	**0.002**			
Test of significance of all canonical axes	**0.002**			

Bold characters indicate highest correlation

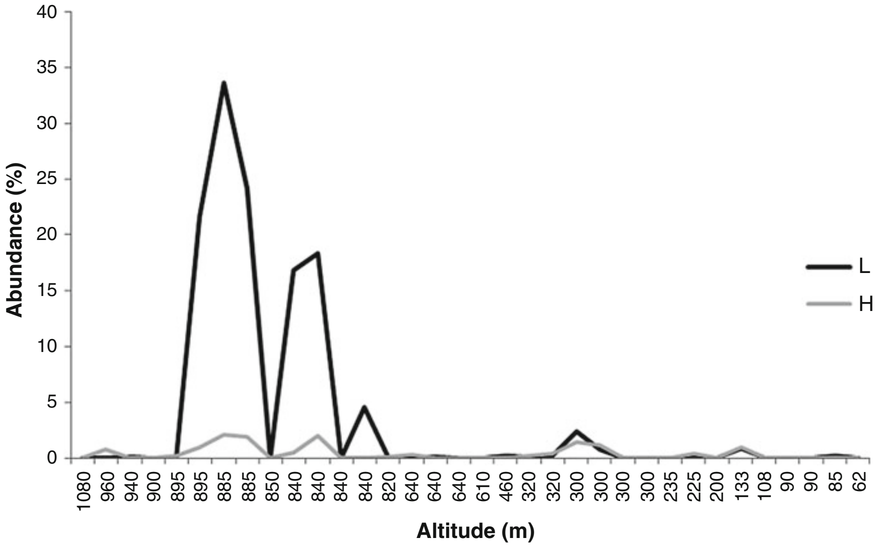


Fig. 3 Abundance (%) of Sericostomatidae (sensitive to pollution family) in reference sites during the high and low flow period

3.1.1 Seasonality

Taxa sensitive and tolerant to pollution were found in all altitudes but their abundance differed (Figs. 3 and 4). Sensitive to organic pollution, benthic macroinvertebrate taxa (e.g. Leuctridae, Ephemeridae, Sericostomatidae, Athericidae)

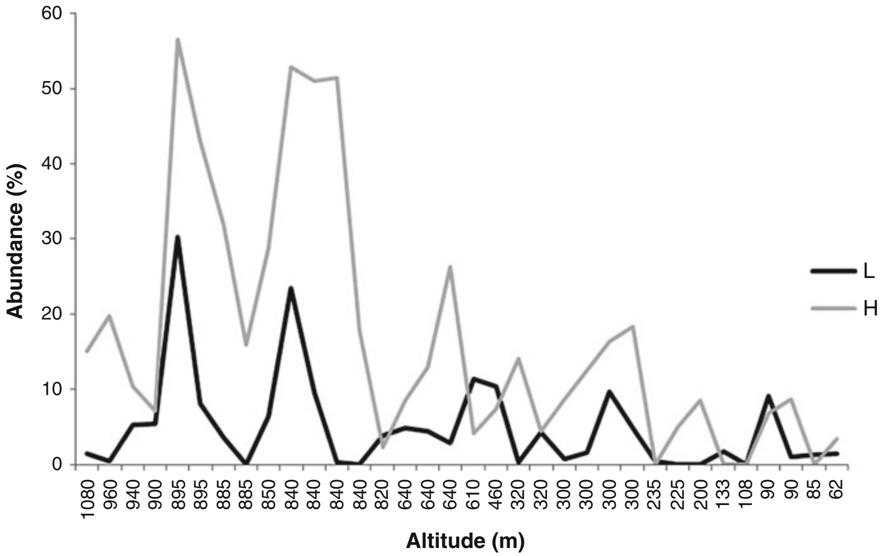


Fig. 4 Abundance (%) of Baetidae (tolerant to pollution family) in reference sites during the high and low flow period

were more abundant during the low flow period in mid and high altitudes (Fig. 3). Tolerant to organic pollution, benthic macroinvertebrate taxa (e.g. Chironomidae, Baetidae, Ephemerellidae) were more abundant during the high flow period in all altitudes (Fig. 4).

3.2 *Assessment of the Ecological Quality: Longitudinal Degradation*

The database of the benthic macroinvertebrates consisted of 833 samples from northern and central Greece from 33 different river basins. In Fig. 5, the distribution of the samples in the five Mediterranean river types is illustrated. Most of the samples belonged to R-M3 and R-M1 type. The results of the STAR_ICMi and HESY indices coincided when the ecological quality was lower than good (Fig. 6). Greater differences existed between good and excellent water quality in the R-M1 sites (Fig. 6). In the R-M4 sites, there were minor differences. According to the assessment based on HESY, the R-M1 sites with a lower than good quality consisted of 21% moderate, 20% poor and 6% bad water quality. In the R-M2 sites, 28% refers to moderate, 41% to poor and 6% to bad water quality. In the R-M4 sites, 6% of the studied sites concern moderate, 1% poor and the rest good or high water quality. The water quality according to HESY differentiates according to

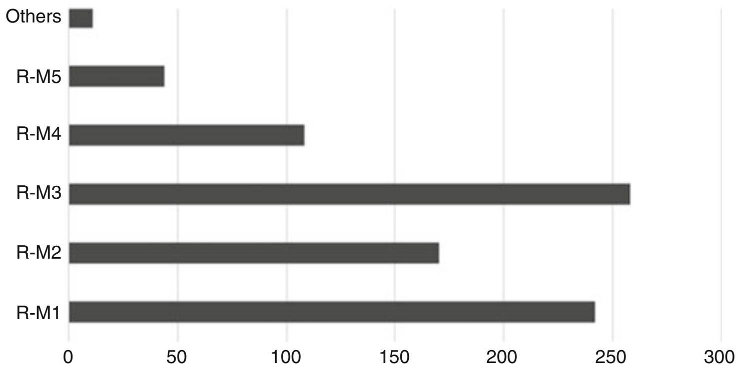


Fig. 5 The distribution of the 833 samples to Mediterranean river types (others: sites derived from catchment areas lower than 10 km^2)

the altitude, with better quality in semi-mountainous (>600 m) and hilly (150–600 m) sites (59% and 60%, respectively, of the sites had good to high water quality) than in the lowland. Only 21% of the lowland sites (<150 m) displayed good to high water quality, and 52% of them had the mostly modified habitats (HMS>8). Semi-mountainous and hilly sites are mostly (80%) unmodified (HMS<8).

In the benthic macroinvertebrate database, 650 sites have been recognised as natural water bodies and 146 as heavily modified water bodies (HMWB). The ecological status of the natural water bodies was classified as of high status for 15 of them, good for 165, moderate for 316, poor for 131 and bad for 13. The ecological potential of HMWB sites was found good in three sites, moderate in 57, poor in 66 and bad in 20.

Finally, when linear regression analysis was applied between SemiHES results and the site distance from the source in the main course of the river basins (except for R-M3), a significant degradation was found longitudinally (e.g. Olynthios, $r^2 = 0.61$, number of sites $(N) = 7$, $p < 0.05$; Sofaditis, $r^2 = 0.64$, $N = 10$, $p < 0.05$).

3.3 IMPRESS Analysis, Risk Assessment and DPSIR Framework

According to the IMPRESS analysis, in most cases (62.5%, Table 3), the immission loads produced in 24 studied basins are higher than the permitted limits for fish life [79]. In 50% of the cases, however, the immissions of all three pollutants exceeded these limits (Table 3). BOD immissions of livestock wastes were the most polluting activity (Fig. 7a). Total nitrogen immissions (Fig. 7b) were mainly due to agricultural use, while most of the phosphorus immissions came from the domestic wastewater and diffuse sources of pollution (Fig. 7c). From the 23 studied basins,

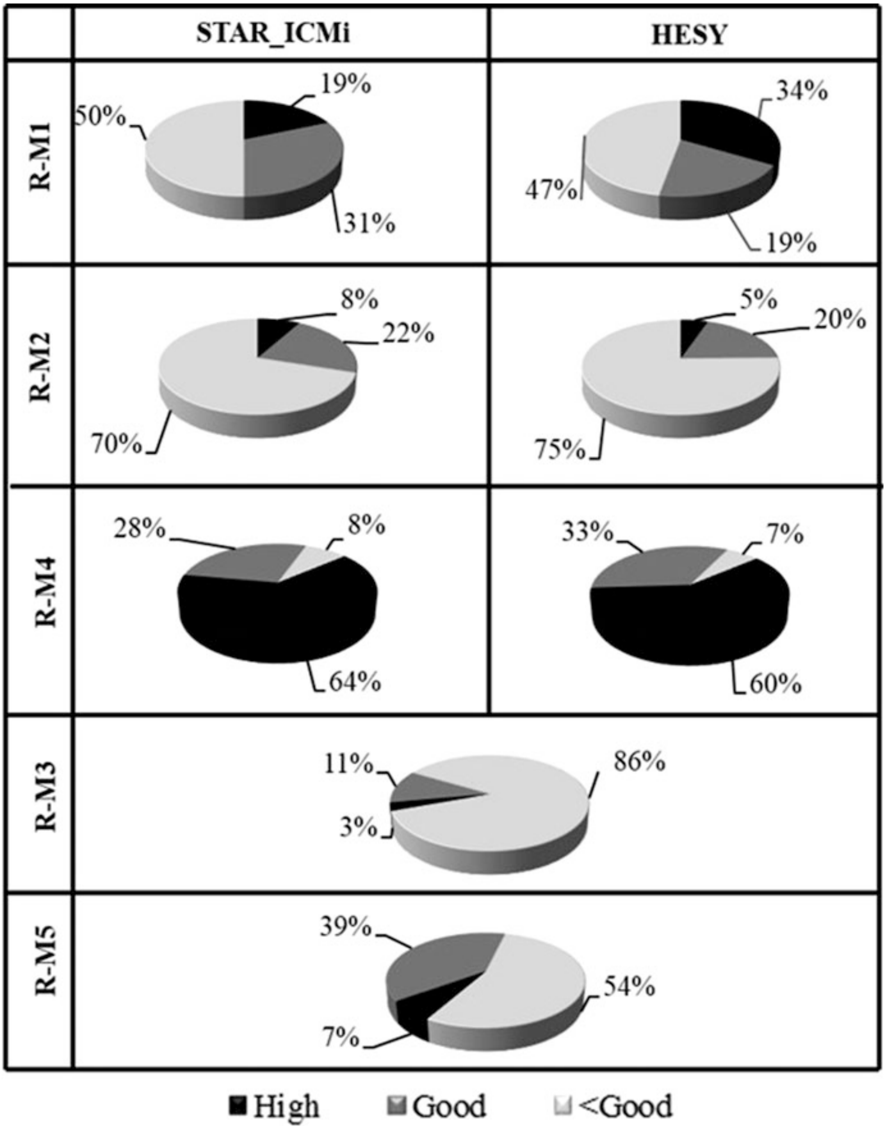


Fig. 6 Water quality according to the European multimetric index STAR_ICMi and the Hellenic Evaluation System in the five intercalibration types

18 of them exhibited moderate water quality, and among them, six need appliance of operational monitoring and six a continuous programme of measures, four surveillance monitoring and the remaining two immediate application of the programme of measures or additional IMPRESS analysis or long-term programme of measures. From the 23 basins, eight of them had mostly modified habitats, six of them need operational monitoring or a continuous programme of measures, and the

Table 3 Comparison of immission loads for 24 studied basins as to the permitted limits for fish life (2006/45/EU)

Basins	BOD	TN	TP
Pineios	+	+	+
Aliakmonas	+	+	+
Axios	+	+	+
Nestos	–	–	–
Almopaios	+	–	+
Chavrias	+	+	+
Olynthios	+	+	+
Pieria	+	+	+
Mavroneri	–	–	–
Kosynthos	+[a]	–	–
Travos	+	–	+
Kompsatos	–	–	–
Sofaditis	+	+	+
Ziliana	–	–	–
Volvi	+	+	+
Melissourgos	+	+	+
Apollonia	–	–	–
Streams of Kastoria Lake, Krepeni	Not applicable because no available data for the flow		
Streams of Kastoria Lake, Toixios	–	+	–
Streams of Kastoria Lake, Xiropotamos	+[a]	+	–
Streams of Kastoria Lake, Vissinias	–	–	–
Streams of Samothraki Island, Xiropotamos	+	–	+[a]
Streams of Samothraki Island, Tsivdogiannis	+	–	–
Streams of Samothraki Island, Fonias	+	–	–

The symbol + means the exceeding of limits
[a]Immission loads exceeding the limit for Salmonidae

rest need surveillance monitoring or immediate application of programme of measures.

According to the DPSIR, the main possible ‘drivers’ affecting the quality of the 23 studied basins were found to be agriculture, livestock and sewage untreated effluents. The ‘pressures’ of these drivers were the fertilisers and pesticides, livestock and domestic wastes. The ‘state’ was affected by the nutrients’ concentration, habitat alterations and changes in the zoobenthos community as to the tolerance of taxa. The ‘impacts’ of the above pressures were the deterioration of ecological water quality and habitat degradation. Finally, the ‘response’ proposed for the improvement of the quality was the change of agricultural policies, management plans for the diffuse pollution loads, treatment of the point pollution loads (domestic, livestock and/or industrial wastes) and connection of the semiurban settlements with wastewater treatment plants (WWTP) according to existing relevant national laws.

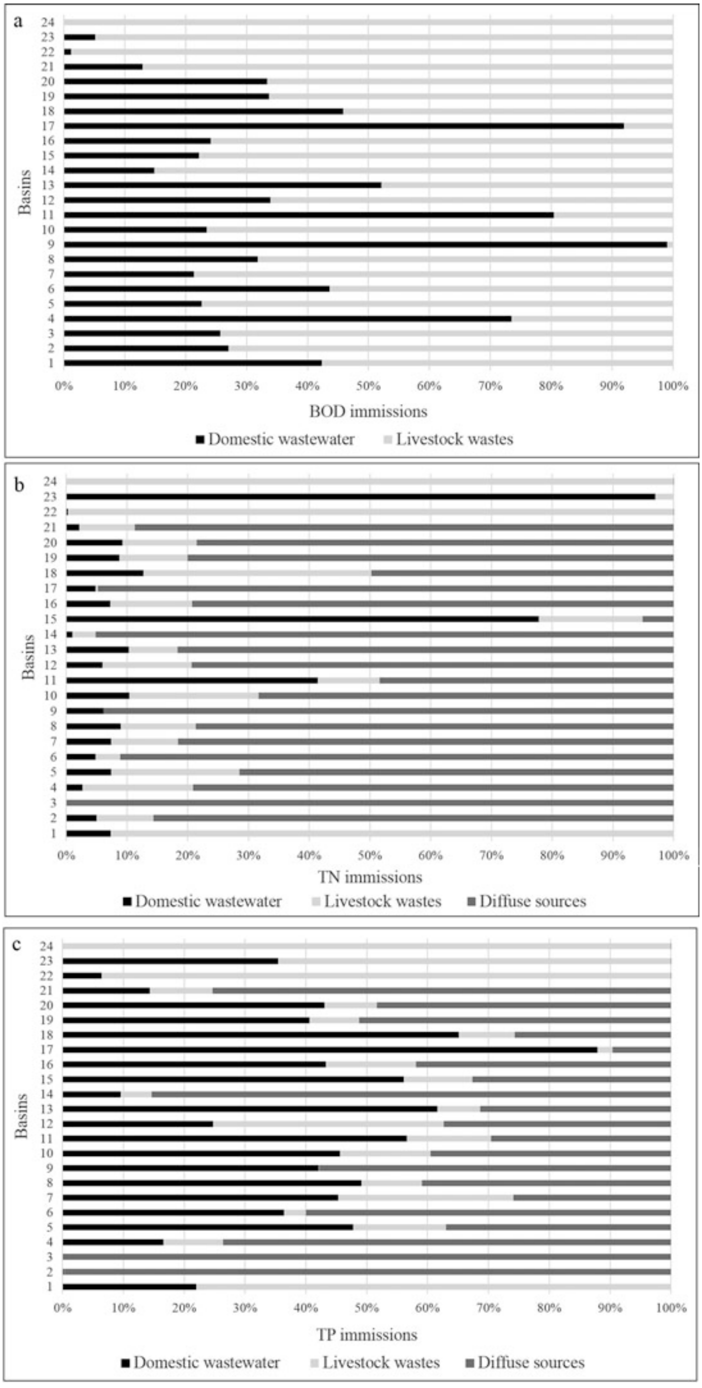


Fig. 7 BOD (**a**), TN (**b**) and TP (**c**) immissions from 24 studied basins produced from domestic wastewater, livestock wastes and diffuse sources (land use). (*1*) Pineios (in this case study livestock wastes and diffuse sources were combined); (*2*) Aliakmonas; (*3*) Axios (in this case

4 Discussion

4.1 Intercalibration River Types and Reference Sites

During the WFD's implementation phase, a wide variety of methodologies have been applied in order to identify reference conditions [94] and achieve the requirements of the intercalibration exercise (IE). The latter is a significant technical and scientific challenge, since it foresees the comparison of method classifications and boundaries applied to national datasets accepting, at the same time, differences in data processing and assessment methods [95] (Fig. 8). Such an intercalibration approach had never been attempted, even in regions where biomonitoring had been applied for a long time (e.g. the USA) [95]. The most crucial step in the intercalibration exercise was the selection of reference sites, since biomonitoring assessment and the development of biological indices or scores are based on their type-specific conditions.

The total number of the reference sites/samples in the northern and central Hellenic reference database is relatively higher in relation to the reference sites selected by all member states for the Mediterranean intercalibration exercise ([69], Table 4). Additionally, physico-chemical parameters of all reference samples had lower values than those proposed for Spain by Bonada et al. [72] and Munné et al. [73]. However, our data were not included in the Mediterranean intercalibration exercise for the boundaries of the H/G and G/M quality of the Hellenic river types (R-M). Reference samples were also in accordance with the levels of morphological alterations (land use) recommended for rivers in Spain, by Sánchez-Montoya et al. [71], and Mediterranean rivers by Feio et al. [96]. However, P-PO_4^{3-} values were higher than those proposed by Feio et al. [96].

In the European Decision 2013/480, the R-M3 type is not included because of the absence of reference sites in lowland regions. Different approaches, however, like the use of historical data and/or the construction of models or expert judgement [97–99] may replace the absence of anthropogenically undisturbed sites in the R-M3 type [94].

In the European Decision 2013/480, the R-M5 type is characterised only by the temporary flow regime. This raises the question of whether intermittent and ephemeral streams in a Mediterranean or European scale are to be considered as the same type in terms of ecological quality assessment, especially towards the WFD

Fig. 7 (continued) study, the loads were estimated for the Greek territory except for the diffuse sources); (*4*) Nestos (in this case study, the loads were estimated only for the upper part of the basin of the Greek territory except for the diffuse sources); (*5*) Almopaios; (*6*) Chavrias; (*7*) Olynthios; (*8*) Pieria; (*9*) Mavroneri; (*10*) Kosynthos; (*11*) Travos; (*12*) Kompsatos; (*13*) Sofaditis; (*14*) Ziliana; (*15*) Volvi; (*16*) Melissourgos; (*17*) Apollonia; (*18*) streams of Kastoria Lake, Krepeni; (*19*) streams of Kastoria Lake, Toixios; (*20*) streams of Kastoria Lake, Xiropotamos; (*21*) streams of Kastoria Lake, Vissinias; (*22*) streams of Samothraki Island, Xiropotamos; (*23*) streams of Samothraki Island, Tsivdogiannis; (*24*) streams of Samothraki Island, Fonias

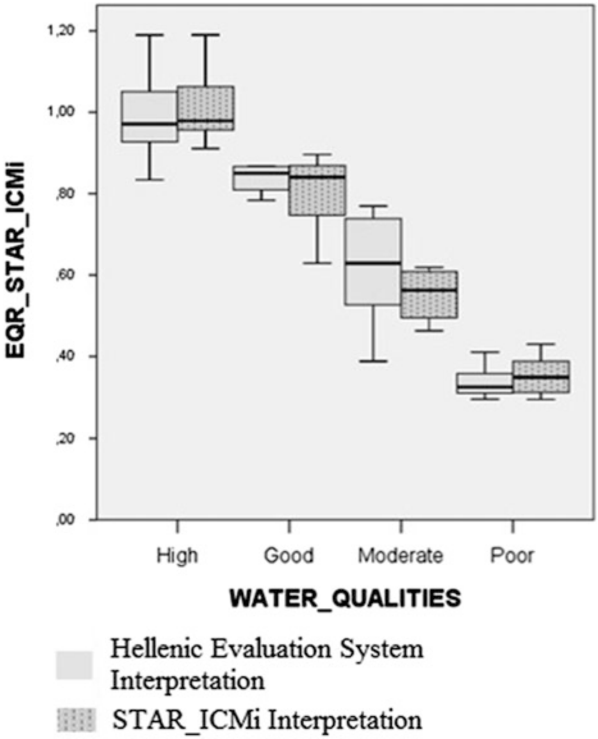


Fig. 8 Range of EQR_STAR_ICMi values in the quality categories defined according to the Hellenic Evaluation System and the STAR_ICMi interpretation in R-M1 river types [70]

Table 4 Comparison of the number of reference sites in the northern and central Hellenic reference database to the Mediterranean intercalibration one [69] by MS

	Northern and central Hellenic reference database		Mediterranean intercalibration exercise	
River type	Sites	Samples	Sites	Samples
R-M1	9	30	80	103
R-M2	10	16	36	41
R-M3	1	1	–	–
R-M4	9	34	48	76
R-M5	7	14	46	65

ecological assessment approach [29]. Regarding the different indices applied by Argyroudi et al. [29], during the low flow season, only the Hellenic Evaluation System could overlook the seasonal variability and assess the ecological quality either as high or good, whereas the rest of the indices applied characterised one of the samples as moderate. Feio et al. [95] and Munné and Prat [100] also reported that a low accuracy in the prediction of ICMi values is achieved for temporary rivers compared to their national assessment approach. This may be caused by the high community biodiversity in these river types as well by the different responses of the EPT taxa to the flow regime [95, 100–102] which is related to the changes between dry and wet periods. For this river type (intermittent and ephemeral streams), as well as the R-M3, in-depth research both at a national and/or Mediterranean level is needed.

4.1.1 Seasonality of Macroinvertebrate Taxa in Reference Sites

It is common to observe a zoobenthos taxon variation as to their distribution and abundance concerning flow, substrate composition, vegetation cover, temperature, stream discharge, etc., but seasonality is one of the prominent causes of temporal variation in the benthic macroinvertebrate assemblages [103]. In this paper, taxa tolerant to pollution were more abundant during the high flow period in high altitudes but showed no seasonality. Sensitive to pollution, taxa, though, showed seasonality in relation to the altitude. Chatzinikolaou et al. [20] and Krno et al. [104] have found Leuctridae to be the most abundant family during low flow periods in high-altitude sites. Moreover, the family Ephemeridae shows distinct preferences in their distribution by being absent from lower altitudes and streams in the islands [105], while it is more common at the mid- and upper-reach sites [106]. Also, this family shows preference for areas where the velocity of the flow is slow because they cannot tolerate fast waters [107–109]. Furthermore, the family Sericostomatidae (Trichoptera) is mostly found in the middle and upper part of the basin [110–112].

4.2 Assessment of the Ecological Quality: Longitudinal Degradation

The ecological quality of the rivers in northern and central Greece is mostly characterised with lower than good quality based on the multimetric index STAR_ICMi and the HESY, especially in the lowland regions. The Mediterranean mountain streams (R-M4) have a better water quality in Greece (93% of the studied sites ranged from good to high quality) than sites in large lowland rivers (R-M3) (86% of the studied sites showed moderate water quality). The same was established for the GES and GEP characterisation. In Europe, the results of the first river management plans (RBMPs), submitted to the European Commission in 2009, indicate the demand for substantial efforts in restoration, because almost 60% of European rivers fail to achieve the WFD good status targets [113]. The downstream section of the rivers is heavily impacted by urbanisation and agriculture, as well as excessive nutrient load affecting the quality with the presence of high abundance of tolerant macroinvertebrate taxa [114, 115]. Affected by the aforementioned conditions, sensitive taxa, represented mainly by Ephemeroptera, Plecoptera and Trichoptera, usually disappear [116]. Moreover, it is important to understand that climate change may bring a further deterioration in water quality on existing resources [117]. River flow systems in the Mediterranean climate zone are highly variable [118]. According to Schneider et al. [119], in the future, the Mediterranean river flows are likely to be even more intermittent, with an increasing number of zero-flow events, thereby creating isolated pools; this will be further sharpened because large amounts of water will also be withdrawn for irrigation purposes [120]. The above will negatively affect the water quality as the concentration of pollutants increases when the flow is reduced [101, 119, 121, 122]. The latter will be accentuated in 'heavily modified' and 'artificial' water bodies having a lower ecological potential as a result of hydromorphological pressures with the risk that the water bodies will not meet the expected targets of the WFD and the need to implement extra measures [123, 124].

A significant proportion of European water bodies have been designated as HMWB. In the Netherlands, Belgium, Slovak Republic and Czech Republic, more than 50% of the water bodies were characterised as HMWB [124]. The rest of the MS have provisionally identified around 16% of their water bodies as heavily modified or artificial [124]. In Germany, 15.5% of the natural water bodies are classified with at least 'good ecological status' and the rest (84.5%) are assessed as 'moderate' (32.5%), 'poor' (30.5%) and 'bad' (4.5%), while only 5% of all heavily modified or artificial water bodies reach the WFD's objectives [125]. In northern and central Greece, 81.7% are natural and among these, 70.8% have lower than good quality. Among heavily modified sites, 97.9% appears to be less than good.

The river continuum concept in undisturbed rivers suggests three sectors longitudinally (upstream, middle and downstream) consisting of distinct macroinvertebrate assemblages [126]. However, a longitudinal degradation in water quality is noted and attributed by several researchers to agricultural and urban activities [114, 127–129]. Longitudinal degradation is recognised as an important factor in structuring macroinvertebrate communities in the rivers [130, 131]. A positive relation was found in the Tripotamos River, when the distance of each site from the confluence site was regressed against the Hellenic Evaluation System values (SemiHes) [57]. The same was found in this study for the Olynthios and Sofaditis Rivers. The absence of longitudinal gradient in basins larger than 1,000 km^2 is explained by the confluence of many tributaries which alter the water quality. The latter is also true for the Pineios River [36]. Similarly, the Nevėžis River is in the middle Lithuanian lowland, and its catchment size is 6,140.5 km^2, where there is no longitudinal gradient [132].

In this study, the hydromorphological changes are increased in lowland regions and have obvious consequences in the loss of naturalness of streams and rivers. According to Feio et al. [96], the hydromorphological changes of human origin especially were almost always present in seven countries in the Mediterranean region, resulting in the low number of selective reference sites. Channelisation, bank alteration and changes in riparian vegetation affect the majority of small streams, and on the other hand, streams in medium-sized catchment are more affected by damming to retain water for power production, fishing and leisure areas [96]. These alterations have expectable consequences for the aquatic biocommunities because they lead to loss of habitat for feeding, reproduction or protection of aquatic animals and loss of retentive capacity for allochthonous inputs [96, 133, 134].

4.3 IMPRESS Analysis, Risk Assessment and DPSIR Model

The European Environment Agency [135] stated that the water bodies are under threat due to excessive nutrient input from point and diffuse sources. In many MS, the high concentration of the nitrogen inputs from agricultural land is a common problem; in the EU, the contribution of agriculture to pollution of surface waters is estimated to be 55% [136]. In Greece, approximately 84% of the river basins had lower than good water quality, and half of these need operational monitoring and the rest a continuous programme of measures. According to the Commission of the European Communities [137], most water bodies of the MS risk failing to meet the environmental objectives of the WFD; MS focused more on operational than on surveillance monitoring (17 of 25 MS) in order to establish the actual status of their water bodies. According to Hering et al. [124], the operational monitoring does not reveal long-term trends, which are independent of the local situation, in contrast to the surveillance monitoring, which is validating the impact assessment, assessing long-term changes in the river basin district and providing information for the design of operational monitoring programmes.

The DPSIR framework comprises a systematic approach to environmental management by exploring the interdisciplinary links [10]. In Mediterranean regions, the main ‘drivers’ are the high population density and agricultural and industrial activities, with ‘pressures’ such as dam constructions for water withdrawal and inputs from not fully treated urban and industrial wastewater plants which represent a serious threat to the biological integrity and diversity (‘impacts’) [114, 129, 138, 139]. In the current study, the main ‘drivers’ affecting the quality of the 23 studied basins were also found to be agriculture, livestock and sewage untreated effluents. According to Tscherning et al. [140], most of the analysed DPSIR model studies addressed the administrative systems, and only few (8 of 21 studies) integrated decision makers in the participative process. It is very difficult for the practitioners of river restoration and decision makers to understand the relation of the pollutants emitted to a river basin by the human activities [10]. Thus, Song and Frostell [10] proposed the use of a simpler model: the DPR (driver-pressure-responses) in order to improve the understanding and management of human pressures on water systems and to develop more proactive strategies and realistic objective systems for water management.

Acknowledgements This paper comprises case studies completed during the past 21 years, which were performed by members/students of the Laboratory of Zoology, School of Biology and the Interdisciplinary Postgraduate Programme at Aristotle University of Thessaloniki entitled ‘Ecological Quality and Management at a basin level’, funded in some cases by national or European sources (http://users.auth.gr/~mlazarid/eng/eng_index.htm).

References

1. Logan P, Furse M (2002) Preparing for the European Water Framework Directive – making the links between habitat and aquatic biota. Aquat Conserv Mar Freshw Ecosyst 12:425–437
2. Borja A, Miles A, Occhipinti-Ambrogi A, Berg T (2009) Current status of macroinvertebrate methods used for assessing the quality of European marine waters: implementing the Water Framework Directive. Hydrobiologia 633:181–196
3. Armitage PD, Moss D, Wright JF et al (1983) The performance of a new biological water quality score system based on macroinvertebrates over a wide range of unpolluted running-water sites. Water Res 17:333–347
4. Alba-Tercedor J, Jáimez-Cuéllar P, Álvarez M et al (2002) Caracterización del estado ecológico de rios mediterráneos ibéricos mediante el indice IBMWP (antes BMWP'). Limnetica 21:175–185
5. Buffagni A, Erba S, Cazzola M et al (2006) The STAR common metrics approach to the WFD intercalibration process: full application for small, lowland rivers in three European countries. In: The ecological status of European rivers: evaluation and intercalibration of assessment methods. Springer, The Netherlands
6. Artemiadou V, Lazaridou M (2005) Evaluation score and interpretation index for the ecological quality of running waters in Northern and Central Hellas. Environ Monit Assess 110:1–40
7. Borja A, Galparsoro I, Solaun O et al (2006) The European Water Framework Directive and the DPSIR, a methodological approach to assess the risk of failing to achieve good ecological status. Estuar Coast Shelf Sci 66:84–96
8. Alexakis D, Kagalou I, Tsakiris G (2013) Assessment of pressures and impacts on surface water bodies of the mediterranean. Case study: Pamvotis lake, Greece. Environ Earth Sci 70:687–698
9. Kagalou I, Leonardos I, Anastasiadou C et al (2012) The DPSIR approach for an integrated river management framework. A preliminary application on a Mediterranean site (Kalamas River -NW Greece). Water Resour Manag 26:1677–1692
10. Song X, Frostell B (2012) The DPSIR framework and a pressure-oriented water quality monitoring approach to ecological river restoration. Water 4:670–682
11. Tsiaousi V (1995) Evaluation of the running water quality and of the applicability of biotic indices in Aggitis river with the use of benthic macroinvertebrates. M.Sc. thesis, Aristotle University of Thessaloniki, Greece
12. Artemiadou V (1996) Monthly fluctuation of water quality in the river Aliakmon (at two sites). Dissertation thesis, Aristotle University of Thessaloniki, Greece
13. Yfantis Y (1996) Ecological evaluation of water quality in the river Aliakmonas at twenty sites in April. Dissertation thesis, Aristotle University of Thessaloniki, Greece
14. Copeland RS, Lazaridou-Dimitriadou M, Artemiadou V et al (1997) Ecological quality of the water in the catchment of river Aliakmonas (Macedonia, Hellas). In: Proceedings of the 5th international congress of environmental science and technology. 2, 1–4
15. Yfantis G, Artemiadou V, Lazaridou-Dimitriadou M et al (1999) Ecological evaluation of water quality in the River Aliakmonas (Macedonia, Hellas). In: Contributions to the zoogeography and ecology of the eastern Mediterranean region, 1, pp 485–493
16. Lazaridou-Dimitriadou M, Artemiadou V, Yfantis G et al (2000) Contribution to the ecological quality of Aliakmon river (Macedonia, Greece): a multivariate approach. Hydrobiologia 410:47–58
17. Somataridou V, Christoforidi P, Domvri K et al (2008) Ecological study of the running waters of Aliakmonas river catchment area according to the WFD during June and July of 2008. In: 3rd Panhellenic Congress on the Panhellenic Union of Bioscientists, Thessaloniki, Greece, 26–28 September 2008
18. Iliadis M, Ioakeimidou K, Kallikazarou N, et al (2010) Ecological quality of running waters of the catchment area of Almopaios river according to the Directive 2000/60/EC during June–

May 2009. In: 5th Pan-Hellenic Congress of the Hellenic Associations of Ecology, Zoology and Botany, Patra, 7–10 October 2010

19. Mergou FE, Lazaridou M, Albanakis K (2012) The effects of dams on rivers ‘continuum’. In: Conference protection and restoration of the environment XI, Thessaloniki, Greece, 3–6 July, pp 1583–1592
20. Chatzinikolaou Y, Dakos V, Lazaridou M (2008) Assessing the ecological integrity of a major transboundary Mediterranean river based on environmental habitat variables and benthic macroinvertebrates (Aoos-Vjose River, Greece-Albania). Int Rev Hydrobiol 93:73–87
21. Chatzinikolaou Y, Dakos V, Lazaridou M (2006) Longitudinal impacts of anthropogenic pressures on benthic macroinvertebrate assemblages in a large transboundary Mediterranean river during the low flow period. Acta Hydrochim Hydrobiol 34:453–463
22. Langrick JM, Artemiadou V, Yfantis G et al (1998) An integrated water quality assessment of the river Axios, Northern Greece. In: Proceedings of the international conference, protection and restoration of the environment IV, Chalkidiki, Greece, 1–4 July 1998, pp 136–143
23. Artemiadou V, Kampa E, Lazaridou-Dimitriadou M (1999) Estimation of the ecological quality of surface waters of the river Axios (Central Macedonia) during 1997. In: 8th European Ecological Congress. The European dimension in ecology, Porto-Kara, 18–23 September 1999
24. Drouin S, Artemiadou V, Lazaridou-Dimitriadou M et al (1999) An integrated water quality assessment of the river Axios during the 1998 low flow season. In: Proceedings of the VI international congress of environmental science and technology, Samos, 30 August–2 September, pp 120–129
25. Kampa E, Artemiadou V, Lazaridou-Dimitriadou M (2000) Ecological quality of the river Axios (N. Greece) during spring and summer 1997. Belg J Zool 130:23–29
26. Chatzinikolaou Y (2001). Monitoring of the ecological water quality and the habitat structure of Axios river. M.Sc. thesis, Aristotle University of Thessaloniki, Greece
27. Fragoulidou T, Koukidou E, Mathioudaki M et al (2008) Ecological study of the running waters of Axios river catchment area according to the WFD during June and July of 2008. In: 3rd Panhellenic Congress on the Panhellenic Union of Bioscientists, Thessaloniki, 26–28 September 2008
28. Simeliadou E (2012) Assessing the ecological water quality in the Chavrias river the year 2011. M.Sc. thesis, Aristotle University of Thessaloniki, Greece
29. Argyroudi A, Chatzinikolaou Y, Poirazidis K et al (2009) Do intermittent and ephemeral Mediterranean rivers belong to the same river type? Aquat Ecol 43:465–476
30. Argyroudi A (2007) Spatial and temporal differences between mediterranean intermittent and ephemeral rivers and impacts of the minor dams of Dadia National Park. Dissertation thesis, Aristotle University of Thessaloniki, Greece
31. Ntislidou Ch, Basdeki A, Papacharalampou Ch et al (2012) Ecological water quality and management at a river basin level. Case study: river basin Kosynthos. In: Conference protection and restoration of the environment XI, Thessaloniki, Greece, 3–6 July 2012, pp 1555–1564
32. Argyriou KA, Avtzi H, Polyxroniadis K et al (2010) Ecological water quality at basin level: case study of Mavroneri, Pieria. In: Proceedings of the 5th Pan-Hellenic Congress of the Hellenic Associations of Ecology, Zoology and Botany, titled “Ecological processes in space and time”, Patra, 7–10 October 2010
33. Margaritis M, Dalaoutzis P (2011) Ecological water quality and management at a river basin level. Case study of Olynthios basin. Interdisciplinary Postgraduate Program, Aristotle University of Thessaloniki, Greece
34. Patsia A, Kanli L, Agelakou G et al (2008) Ecological study of running waters of the Greek part of Nestos catchment area according to the WFD in June 2008. In: 3rd Panhellenic Congress on the Panhellenic Union of Bioscientists, Thessaloniki, Greece

35. Prieto Montes M, Patsia A, Angelakou G et al (2009). Ecological quality of river Nestos (Hellas) and its tributaries in September 2008. In: Proceedings of the 11th international congress on the zoogeography and ecology of Greece and adjacent regions, Irakleio, 25–29 September 2009
36. Chatzinikolaou Y (2008) Effect of management practices on the water quality and ecology of rivers in Greece. Pinios river as a study case. Ph.D. thesis, Aristotle University of Thessaloniki, Greece
37. Betziou M, Pana E, Patetsini E (2010) Ecological water quality and management at a river basin level. Case study of Pieria Basin. Interdisciplinary Postgraduate Program, Aristotle University of Thessaloniki, Greece
38. Katsikatsou M (2006) Study at the upper reaches of reference sites in Pinios catchment area. Dissertation thesis, Aristotle University of Thessaloniki, Greece
39. Katsikatsou M, Sánchez-Montoya MM, Ilia G et al (2006) Study at the upper reaches of reference sites in Pinios catchment area. In: Proceedings of the 10th international congress on the zoogeography and ecology of Greece and adjacent regions, Patra, 26–30 June 2006, p 150
40. Ford J, Yfantis G, Artemiadou V et al (1998) Ecological evaluation of water quality in river Mavrolakkas (Olympiada, Halkidiki) from May to August 1998. In: Proceedings of the international conference 'Protection and restoration of the environment IV'. 1, pp 144–152
41. Lazaridou-Dimitriadou M, Koukoumides H, Lekka M et al (2004) Integrative evaluation of the ecological quality of metalliferous streams (Chalkidiki, Macedonia, Hellas). Environ Monit Assess 91:59–86
42. Lampou A (2012) Assessment of the ecological water quality, according to the Directive 2000/60/EC, in rivers of Samothraki island in 2011. M.Sc. thesis, Aristotle University of Thessaloniki, Greece
43. Nikolopoulou P (2010) Ecological situation and impress analysis in Sofaditis river basin. M. Sc. thesis, Aristotle University of Thessaloniki, Greece
44. Theofanoudi A, Vourka A, Kougioumtzidou K et al (2010) Assessing ecological quality of the surface water bodies of Sofaditis river basin (Perf. Karditsa) according to the Water Framework Directive 2000/60/EC. In: Proceedings of the 33rd scientific conference of Hellenic society for biological sciences, Edessa, 19–21 May 2011, pp 92–93
45. Ilia G, Ioannou A, Patsia A et al (2005) Assessment of the water and habitat quality of the Greek part of Strymonas river (Macedonia, Hellas), in accordance with the EU Water Framework Directive. IASME Trans 6:1059–1064
46. Patsia A (2006) Ecological evaluation of the water quality of the Strymonas river upstream and downstream of the Lithotopos dam. Dissertation thesis, Aristotle University of Thessaloniki, Greece
47. Patsia A (2009) Photographic key characters of the Hellenic evaluation system taxa of benthic macroinvertebrates. Case studies: Strymonas and Nestos rivers. M.Sc. thesis, Aristotle University of Thessaloniki, Greece
48. Patsia A, Lazaridou M (2010) Comparison of ecological quality based on benthic macroinvertebrates at same river types at the Greek parts of two transboundary catchment areas of N. Greece (Strymonas and Nestos rivers). In: 5th Pan-Hellenic Congress of the Hellenic Associations of Ecology, Zoology and Botany, Patra, 7–10 October 2010
49. Zabour G, Nikolopoulou P, Katikaridis G et al (2010) Ecological quality of Richios river and Appolonia stream according to the Directive 2000/60/EC during June and October 2009. In: 5th Pan-Hellenic Congress of the Hellenic Associations of Ecology, Zoology and Botany, titled "Ecological processes in space and time", Patra, 7–10 October 2010
50. Mergou F, Stavrakidou E (2011) Ecological water quality at a river basin level. Case study of streams Bogdanas, Sxolari and Lagadikia. Interdisciplinary Postgraduate Program, Aristotle University of Thessaloniki, Greece
51. Vavalidis T (2011) Assessing the ecological water quality with benthic macroinvertebrates and fish in the rivers of NeaApollonia and Melissourgos. M.Sc. thesis, Aristotle University of Thessaloniki, Greece

52. Vavalidis T, Bobori CD, Lazaridou M (2011) Assessing pressure drivers on benthic macroinvertebrate and fish communities: a case study from two small Mediterranean rivers. International Scientific Session “The human impact on biodiversity in Black Sea”, Agigea – Constanta, Romania, 21–22 October 2011
53. Artemiadou V, Statiri X, Brouziotis T et al (2008) Ecological quality of small mountainous Mediterranean streams (river type R-M4) and performance of the European intercalibration metrics. Hydrobiologia 605:75–88
54. Artemiadou V, Statiri X, Brouziotis Th, Lazaridou M (2006) Water quality evaluation of small-sized mountainous Mediterranean streams (RM4) and performance of the European Intercalibration metrics. In: Proceedings of the 10th international congress on the zoogeography and ecology of Greece and adjacent regions, Patra, 26–30 June 2006
55. Artemiadou V (2007) Predictive model for the assessment of running water quality at the basin of Tavropos Reservoir using benthic macroinvertebrates and the Hellenic Evaluation System. Ph.D. thesis, Aristotle University of Thessaloniki, Greece
56. Navrozidou V, Kostinaki D (2012) Ecological water quality and management at a river basin level. Case study of Vistonida basin. Interdisciplinary Postgraduate Program, Aristotle University of Thessaloniki, Greece
57. Cheimonopoulou Th M, Bobori CD, Theocharopoulos I et al (2011) Assessing ecological water quality with macroinvertebrates and fish: a case study from a small Mediterranean river. Environ Manage 47:279–290
58. Cheimonopoulou Th M (2005) Ecological evaluation of Tripotamos- Aliakmonas river system. M.Sc. thesis, Aristotle University of Thessaloniki, Greece
59. Polychroniadis K (2010) Typology and estimation of ecological condition of basin of Ziliana river with the use of Geographic Information Systems (GIS). M.Sc. thesis, Aristotle University of Thessaloniki, Greece
60. ISO 7828:1985 (BS EN 27828) (1994) Water quality – methods of biological sampling – guidance on handnet sampling of aquatic benthic macro-invertebrates (ISO 7828 1985)
61. Armitage PD, Hogger J (1994) Invertebrates ecology and methods of survey. In: Ward D, Holmes N, Jose P (eds) The new rivers and wildlife handbook. RSPB, Sandy, Bedfordshire
62. Kemitzoglou D (2004) The effectiveness of the semi-quantitative sampling method using benthic macroinvertebrates. Dissertation thesis, Aristotle University of Thessaloniki, Greece
63. Wright JF (2000) An introduction to RIVPACS. In: Wright JF, Sutcliffe DW, Furse MT (eds) Assessing the biological quality of fresh waters: RIVPACS and other techniques. Freshwater Biological Association, Ambleside, pp 1–24
64. APHA (1985) Standard methods for the examination of water and wastewater, 16th edn. American Public Association, Washington, DC
65. Wentworth C (1922) A scale of grade and class terms for clastic sediments. J Geol 30:377–392
66. Raven P, Boon P, Dawson F et al (1998) Towards an integrated approach to classifying and evaluating rivers in the UK. Aquat Conserv 8:383–393
67. Raven P, Holmes N, Dawson F et al (1998) Quality assessment using River Habitat Survey data. Aquat Conserv 8:477–499
68. van de Bund WJ, Cardoso AC, Heiskanen AS et al (2004) Overview of common intercalibration types. Electronic document. http://wfd-reporting.jrc.cec.eu.int/Docs/typesmanual
69. van de Bund WJ (2009) Water Framework Directive intercalibration technical report. Part 1: rivers. JRC scientific and technical reports. ISSN 1018-5593, 179 pp
70. Ntislidou C, Artemiadou V, Kanli L et al (2013) The intercalibration of the Hellenic evaluation system for Mediterranean rivers of type R-M1 and R-M2 at Northern and Central Greece. Ecol Indic 29:208–218
71. Sánchez-Montoya MM, Vidal-Abarca MR, Suárez ML (2005) Propuesta de criterios para la selección de estaciones de referencia en ríos mediterráneos en el contexto de la Directiva Marco del Agua. Tecnología del Agua 266:42–53

72. Bonada N, Prat N, Munné A et al (2002) Criterios para la selección de condiciones de referencia en los ríos mediterráneos. Resultados del proyecto GUADALMED. Limnetica 21:99–114
73. Munné AC, Solà C, Prat N (2006) Estado ecologico de los rios en Cataluña. Diagnosis del riesgo de incumplimiento de los objectivos de la Directiva Marco del Agua. Tecnologia del Agua 26:30–46
74. Ter Braak CJF, Šmilauer P (1998) CANOCO reference manual and user's guide to CANOCO for windows. Software for Canonical Community Ordination (version 4). Centre for Biometry, Wageningen, 351 pp
75. Clarke KR, Gorley RN (2001) Primer v5: user manual/tutorial. Primer-E Ltd, Plymouth, 91 pp
76. Working Group 2A (2003) Overall approach to the classification of ecological status and ecological potential. Office for Official Publications of the European Communities, Luxembourg, 53 pp
77. ETC/ICM (2015) European Freshwater Ecosystem Assessment; cross-walk between the Water Framework Directive and Habitats Directive types, status and pressures, ETC/ICM Technical Report 2/2015. European Topic Centre on Inland, Coastal and Marine Waters, Magdeburg, 95 pp
78. EU (2008) Commission decision of 30 October 2008 establishing, pursuant to Directive 2000/60/EC of the European Parliament and of the Council, the values of the Member State monitoring system classifications as a result of the intercalibration exercise (notified under document number C(2008) 6016) (text with EEA relevance) (2008/915/EC). Off J Eur Communities L332(20):1–25
79. EU (2006) Directive 2006/44/EC of the European Parliament of the Council of 6 September 2006 on the quality of fresh waters needing protection or improvement in order to support fish life. Off J Eur Communities L264(20):1–12
80. Gabriels W, Goethals PLM, De Pauw N (2005) Implications of taxonomic modifications and alien species on biological water quality assessment as exemplified by the Belgian Biotic Index method. Hydrobiologia 542:137–150
81. Buffagni A, Erba S, Birk S, Cazzola M, Feld C, Ofenböck T, Murray-Bligh J, Furse MT, Clarke R, Herring D, Soszka H, van de Bund W (2005) Towards European Inter-Calibration for the Water Framework Directive: procedures and examples for different river types from the E.C. project STAR, Instituto di Ricerca Sulle Acque, Rome, 460 pp
82. Fribourg-Blanc B, Courbet C (2004) Calcul desemissions dé polluants organiques et de fertilisants dans l'eau, Second lot: collecte et mise en forme des informations de base necessaires au calcul desemissions urbaines et industrielles répertoriées, dispersées et surfaciques. Office International de l'Eau, Paris, France
83. Andreadakis A, Gavalakis E, Kaliakatsos L et al (2007) The implementation of the Water Framework Directive (WFD) at the river basin of Anthemountas with emphasis on the pressures and impacts analysis. In: Proceedings of the 9th international conference on environmental science and technology, Rhodes island, Greece
84. Economopoulos AP (1993) Assessment of sources of air, water and land pollution. Guide to rapid source inventory techniques and their use in formulating environmental control strategies. Part one: rapid inventory techniques in environmental pollution. World Health Organization, Geneva
85. Decision 4813/98 (1998) 'Determination of usage of surface waters and special terms for waste disposal in natural receivers at Trikala Prefecture' FEK-575B/11-6-1998
86. LAWA (2002) German Guidance Document for the implementation of the EC Water Framework Directive. Länderarbeitsgemeinschaft Wasser, Mainz, Germany
87. EEA (2005) Technical assessment method for morphological alterations in rivers. Water Framework Directive Programme – Environment Agency, 1–26

88. Castro CD, Infante AP, Rodriguez JR et al (2005) Manual para la identificacion de las presiones y analisis del impacto en aguas superficiales. Ministerio de Medio Ambiente, Secretaría General Técnica, Centro de Publicaciones
89. Working Group 2.1 – IMPRESS (2003) Analysis of pressures and impacts. Office for Official Publications of the European Communities, Luxembourg, 157 pp
90. Spangenberg JH, Douguet JM, Settele J, Heong KL (2015) Escaping the lock-in of continuous insecticide spraying in rice: developing an integrated ecological and socio-political DPSIR analysis. Ecol Model 295:188–195
91. Gari SR, Newton A, Icely JD (2015) A review of the application and evolution of the DPSIR framework with an emphasis on coastal social-ecological systems. Ocean Coast Manag 103:63–77
92. EU (2013) Commission decision of 20 September 2013 establishing, pursuant to Directive 2000/60/EC of the European Parliament and of the council, the values of the member State monitoring system classifications as a result of the intercalibration exercise and repealing decision 2008/915/EC. Off J Eur Communities L266(1):1–47
93. Lazaridou M, Kanli L, Ntislidou Ch et al (2014) Comparison of river typology systems in Northern and Central Greece. EWRA 42:3–18
94. Baattrup-Pedersen A, Søren EL, Riis T (2013) From expert judgement to supervised classification: a new approach to assess ecological status in lowland streams. Sci Total Environ 447:116–122
95. Feio MJ, Ferreira J, Buffagni A et al (2013) Comparability of ecological quality boundaries in the Mediterranean basin using freshwater benthic invertebrates. Statistical options and implications. Sci Total Environ. doi:10.1016/j.scitotenv.2013.07.085
96. Feio MJ, Aguiar FC, Almeida SFP et al (2013) Least disturbed condition for European Mediterranean rivers. Sci Total Environ. doi:10.1016/j.scitotenv.2013.05.056
97. Muxika I, Borja A, Bald J (2007) Using historical data, expert judgement and multivariate analysis in assessing reference conditions and benthic ecological status according to the European Water Framework Directive. Mar Pollut Bull 55:16–29
98. Nijboer RC, Johnson RK, Verdonschot PFM et al (2004) Establishing reference conditions for European streams. In: Integrated assessment of running waters in Europe. Springer, The Netherlands, pp 91–105
99. Vanden Bossche JP, Usseglio-Polatera P (2005) Characterization, ecological status and type-specific reference conditions of surface water bodies in Wallonia (Belgium) using biocenotic metrics based on benthic invertebrate communities. Hydrobiologia 55:253–271
100. Munné A, Prat N (2011) Effects of Mediterranean climate annual variability on stream biological quality assessment using macroinvertebrate communities. Ecol Indic 11:651–662
101. Bonada N, Rieradevall M, Prat N (2007) Macroinvertebrate community structure and biological traits related to flow permanence in a Mediterranean river network. Hydrobiologia 589:91–106
102. Gárcia-Roger EM, Sánchez-Montoya MM, Gómez R et al (2011) Do seasonal changes in habitat features influence aquatic macroinvertebrate assemblages in perennial versus temporary Mediterranean streams? Aquat Sci 73:567–579
103. Linke S, Bailey CR, Schwindt J (1999) Temporal variability of stream bioassessments using benthic macroinvertebrates. Freshw Biol 42:575–584
104. Krno I, Šporka F, Štefková E (2013) The influence of environmental variables on larval growth of stoneflies (Plecoptera) in natural and deforested streams. Biologia 68:950–960
105. Skoulikidis Th N, Karaouzas I, Gritzalis CK (2009) Identifying key environmental variables structuring benthic fauna for establishing abiotic typology for Greek running waters. Limnologica 39:56–66
106. Jiang XM, Xiong J, Qui JW et al (2010) Structure of macroinvertebrate communities in relation to environmental variables in a subtropical Asian river system. Int Rev Hydrobiol 95:42–57

107. Bedford JS (2009) The effects of riparian habitat quality and biological water quality on the European Otter (*Lutra lutra*) in Devon. Biosci Horiz 2:125–133
108. Kim BS, Youn HJ, Seo EW et al (2011) The influence of sediment control dam dredging on benthic macroinvertebrate communities of mountain stream. Entomol Res 41:288–289
109. Tali I, Pir Z, Siddiqui A et al (2013) Analysis of mayflies (Insecta: Ephemeroptera) and physico-chemical properties of river Narmada Madhya Pradesh India. Int J Recent Sci Res 7:1027–1031
110. Bredenhand E, Samways JM (2009) Impact of a dam on benthic macroinvertebrates in a small river in a biodiversity hotspot: Cape Floristic region, South Africa. J Insect Conserv 13:297–307
111. Ferreira V, Graça ASM, Feio MJ et al (2004) Water quality in the Mondego river basin: pollution and habitat heterogeneity. Limnetica 23:295–306
112. Miserendino ML, Pizzolon AL (2003) Distribution of macroinvertebrate assemblages in the Azul-Quemquemtreu river basin, Patagonia, Argentina. N Z J Mar Freshw Res 37:525–539
113. Hering D, Borja A, Carvalho L et al (2013) Assessment and recovery of European water bodies: key messages from the WISER project. Hydrobiologia 704:1–9
114. Manfrin A, Larsen S, Traversetti L et al (2013) Longitudinal variation of macroinvertebrate communities in a Mediterranean river subjected to multiple anthropogenic stressors. Int Rev Hydrobiol 98:155–164
115. Paul MJ, Meyer JL (2001) Streams in the urban landscape. Annu Rev Ecol Syst 32:333–365
116. Ortiz DJ, Puig MA (2007) Point source effects on density, biomass and diversity of benthic macroinvertebrates in a Mediterranean stream. River Res Appl 23:155–170
117. Dallas FH (2013) Ecological status assessment in Mediterranean rivers: complexities and challenges in developing tools for assessing ecological status and defining reference conditions. Hydrobiologia 719:483–507
118. Oueslati O, De Girolamo AM, Abouabdillah A et al (2010) Attempts to flow regime classification and characterisation in mediterranean streams using multi-variate analysis. In: Workshop advances in statistical hydrology, Taormina, Italy, 23–25 May 2010, pp 1–15
119. Schneider C, Laizé CLR, Acreman MC et al (2013) How will climate change modify river flow regimes in Europe? Hydrol Earth Syst Sci 17:325–339
120. Schaldach R, Koch J, Aus der Beek T et al (2012) Current and future irrigation water requirements in pan-Europe: an integrated analysis of socio-economic and climate scenarios. Global Planet Change 94:33–45
121. Avila A, Neal C, Terradas J (1996) Climate change and implications for streamflow and streamwater chemistry in a Mediterranean catchment. J Hydrol 177:99–116
122. Lindhl G (1992) Hydrological and water resources impact of climate change. In: Jeftic L, Milliman JD, Sestini G (eds) Climatic change and the Mediterranean: environmental and societal impacts of climatic changes and sea level rise in the Mediterranean region. Edward Arnold, London, pp 58–93
123. Borja A, Elliott M (2007) What does 'good ecological potential' mean, within the European Water Framework Directive? Mar Pollut Bull 54:1559–1564
124. Hering D, Borja A, Carstensen J et al (2010) The European Water Framework Directive at the age of 10: a critical review of the achievements with recommendations for the future. Sci Total Environ 408:4007–4019
125. Richter S, Völker J, Borchardt D et al (2013) The Water Framework Directive as an approach for Integrated Water Resources Management: results from the experiences in Germany on implementation, and future perspectives. Environ Earth Sci 69:719–728
126. Vannote RL, Minshall GW, Cummins KW et al (1980) The river continuum concept. Can J Fish Aquat Sci 37:130–137
127. Marzin A, Verdonschot FMP, Pont D (2013) The relative influence of catchment, riparian corridor and reach-scale anthropogenic pressures on fish and macroinvertebrate assemblages in French rivers. Hydrobiologia 704:375–388

128. Marziali L, Armanini DG, Cazzola M et al (2010) Responses of Chironomid larvae (insecta, Diptera) to ecological quality in Mediterranean river mesohabitats (South Italy). River Res Appl 26:1036–1051
129. Solimini AG, Benvenuti A, D'Olimpio R et al (2001) Size structure of benthic invertebrate assemblages in a Mediterranean river. J N Am Benthol Soc 20:421–431
130. Hynes HBN (1970) The ecology of running waters. University of Toronto Press, Toronto
131. Leunda PM, Oscoz J, Miranda R et al (2009) Longitudinal and seasonal variation of the benthic macroinvertebrates community and biotic indices in an undisturbed Pyrenean river. Ecol Indic 9:52–63
132. Virbickas T, Pliūraitė V, Kesminas V (2011) Impact of agricultural land use on macroinvertebrate fauna in Lithuania. Pol J Environ Stud 20:1327–1334
133. Muotka T, Paavola R, Haapala A et al (2002) Long-term recovery of stream habitat structure and benthic invertebrate communities from in-stream restoration. Biol Conserv 105:243–253
134. Petersen LBM, Petersen RC (1991) Short term retention properties of channelized and natural streams. Internationale Vereinigung fuer Theoretische und Angewandte Limnologie. Verhandlungen IVTLAP 24(3):1756–1759
135. EEA (2010) The European environment—state and outlook 2010: synthesis. European Environment Agency, Copenhagen, pp 1–228
136. Volk M, Liersch S, Schmidt G (2009) Towards the implementation of the European Water Framework Directive? Lessons learned from water quality simulations in an agricultural watershed. Land Use Policy 26(3):580–588
137. EU (2009) Commission staff working document accompanying the report of the Commission to the European Parliament and the Council in accordance with article 18.3 of the Water Framework Directive 2000/60/EC on programmes for monitoring of water status. Commission of the European Communities 1–6
138. Prat N, Munné A (2000) Water use and quality and stream flow in a Mediterranean stream. Water Res 34:3876–3881
139. Sabater S, Tockner K (2010) Effects of hydrologic alterations on the ecological quality of river ecosystems. In: Water scarcity in the Mediterranean, Handbook environmental chemistry. Springer, Berlin, pp 15–39
140. Tscherning K, Helming K, Krippnera B et al (2012) Does research applying the DPSIR framework support decision making? Land Use Policy 29:102–110

Socio-Economics and Water Management: Revisiting the Contribution of Economics in the Implementation of the Water Framework Directive in Greece

Phoebe Koundouri, Dimitrios Reppas, and Vassilis Skianis

Abstract This chapter sets out the socio-economic principles that should govern water resources management for the achievement of a sustainable allocation of the resource over time and across space, in accordance with the EU Water Framework Directive. The resulting allocation should be economically efficient, socially equitable and acceptable and environmentally sustainable. The main background concept guiding the identification of such an allocation is the 'total economic value (TEV)' of water resources. This concept derives from the ecosystem goods and services that water resources provide the economy and society. In this chapter we present the state of the art with regard to estimating the TEV of water resources and explain how these estimations can facilitate the design and implementation of different European policies in relation to mitigation of different forms of water stress.

Keywords Nonmarket valuation, Total economic value, Water framework directive, Water valuation

P. Koundouri (✉)
Department of International and European Economic Studies, Athens University of Economic and Business, 76 Patission Street, Athens 104 34, Greece

ICRE8 International Centre for Research on the Environment and the Economy, Artemidos 6 and Epidavrou, Maroussi, 15125 Athens, Greece

Grantham Research Institute on Climate Change and the Environment, London School of Economics and Political Science, Houghton Street, London WC2A 2AE, UK
e-mail: pkoundouri@aueb.gr

D. Reppas
ICRE8 International Centre for Research on the Environment and the Economy, Artemidos 6 & Epidavrou, Maroussi, 15125 Athens, Greece

V. Skianis
Department of International and European Economic Studies, Athens University of Economic and Business, 76 Patission Street, Athens 104 34, Greece

N. Skoulikidis et al. (eds.), *The Rivers of Greece: Evolution, Current Status and Perspectives*, Hdb Env Chem (2018) 59: 357–378, DOI 10.1007/698_2015_427,

Contents

Abbreviations

CBA Cost-benefit analysis
CEM Choice experiment method
CV Contingent valuation
RBD River basin district
TEV Total economic value
WFD Water framework directive
WTP Willingness to pay

1 Introduction

In this chapter, we provide a state-of-the-art review of the basic economic valuation methods that can be used for the monetisation of the economic and societal benefits provided by water resources and discuss how the valuation outcomes can inform policymakers for a more efficient water management plan, in accordance with the European Union's Water Framework Directive (WFD) [1]. Contrary to previous pieces of legislation which focused on specific water-related environmental issues, the WFD aimed at creating an integrated policy framework for the sustainable management and protection of aquatic resources (inland surface waters, transitional waters, coastal waters and groundwater) both in terms of quantity and quality across European Union country members [2]. Therefore, as stated in Waternote 9,[1] the Directive has developed a 'combined approach for point and diffuse sources and refers to several related directives' (p. 1). The necessity for the development of such

[1] Waternote 9 can be accessed here (last accessed 12/02/2015): http://ec.europa.eu/environment/water/participation/pdf/waternotes/water_note9_other_water_legislation.pdf.

a policy framework became imperative by taking into account the increased demand for high-quality water quantities. For the implementation of the Directive, all member states are obliged within specific deadlines to identify all individual river basins within their national territories and assign them to specific river basin districts (RBDs).

2 Economic Aspects of Water Framework Directive

Given the increased water scarcity, the WFD has recognised the need of incorporating economic analysis in the water-related policy agenda through the use of appropriate economic instruments for assessing water value, thus meeting certain environmental objectives, in accordance with the various articles of the Directive. Economic issues are mainly discussed in articles 5 and 9 and in Annex III.

According to article 5, all member states need to undertake an analysis of each RBD characteristics, review the impact of human behaviour on the status of surface water and groundwater and proceed with an economic analysis of water use. Although each country shall proceed and implement its own techniques, the European Union's guidelines [3] suggest the following implementation steps: (1) characterise the river basin in terms of the economics of water uses, trends in water supply and demand levels and current recovery levels of water services' costs, (2) identify all waterbodies or groups of waterbodies that fail to meet the environmental objectives of the Directive, and (3) develop appropriate programmes of measures to be included in river basin management plans through a cost-effectiveness analysis and justify potential derogation from an economic perspective.

As highlighted in article 9 and Annex III, countries shall take into account the principle of cost recovery (including environmental and resource costs) of water services and consider the social, environmental and economic effects of the recovery and also the regional geographical and climatic conditions of each RBD. Table 1 provides a summary of the total cost of water services. The goal is to ensure an adequate contribution of the various water users (industry, households and agriculture) to the cost recovery of water services and to provide strong incentives for users to use resources efficiently. It is also crucial to evaluate the cost of the application of various measure programmes and choose the most cost-effective combination.

Overall, according to the relevant EU guidelines [3], the contribution of economic analysis is along the following topics: (1) understanding the importance of economic issues and trade-offs at each river basin; (2) identifying the most cost-effective way (e.g. through water prices, pollution charges or environmental taxes) for achieving certain environmental objectives for water resources, given the limited availability of financial resources; (3) evaluating the role of various measures for the improvement of water status and considering policies for the compensation of losers; and (iv) relaxing the environmental objectives on waterbodies, if this can help promote overall sustainability.

Table 1 Total economic cost of water services

Financial cost	Cost of providing and administering water services. Includes capital cost, operation cost, maintenance cost and administrative cost
Environmental cost	Environmental cost represents the costs of damage that water uses impose on the environment/ecosystems (e.g. a reduction in the ecological quality of aquatic ecosystems or the salinisation and degradation of productive soils)
Resource cost	Resource cost represents the costs of foregone opportunities which other uses suffer due to the depletion rate of recharge or recovery of water (e.g. linked to the over-abstraction of groundwater)

Source: Koundouri et al. [2], p. 10

The following sections discuss how economic analysis has developed a variety of appropriate tools for meeting the demands of the WFD. These tools allow us to quantify the total economic value of aquatic resources and inform policymakers about the effectiveness and sustainability of proposed management actions.

3 Methodology for Implementing the WFD

When a fully functioning market exists, as in the case of private goods, the value of the assessed asset is normally reflected in the market price (e.g. fish products are priced in a market). However, a market value does not exist for services such as recreation activities or biodiversity. In this section, we provide an overview of the most important economic techniques employed for identifying and estimating water's total economic value (or, at least, some components of it).

3.1 Total Economic Value

The total economic value (TEV) comprises two main types of values that can be derived from an environmental resource: use and non-use values. The former refer to benefits that people receive from the usage of the specific commodity, while the latter refer to benefits people attach to the commodity even if they do not make use of it. Use values can be further divided into three main categories: direct use values, arising from the consumptive use of a certain environmental good; indirect use values, arising when individuals indirectly interact with the resource; and option values, representing the potential benefits that can be derived from the environmental asset by future generations. Non-use values can be further classified into existence values, i.e. values individuals place on the existence of the environmental good as it stands; bequest values, i.e. values individuals place on the importance of preserving the environmental asset for future generations; and altruistic values, i.e. values individuals place on the need to maintain an environmental good in order

Table 2 TEV components for water resources

Use values	
Direct use values	Indirect use values
Irrigation for agriculture	Nutrient retention
Domestic and industrial water supply	Pollution abatement
Energy resources (hydroelectric, fuel wood, peat)	Flood control and protection
Transport and navigation	Storm protection
Recreation/amenity	External ecosystem support
	Micro-climatic stabilisation
Option values	Reduced global warming
Potential future uses of direct and indirect uses	Shoreline stabilisation
Future value of information of biodiversity	Soil erosion control
Non-use values	
Biodiversity	
Cultural heritage	
Bequest, existence and altruistic values	

Source: Birol et al. [4], p. 107

to be used by other individuals. Table 2 provides examples of these various components of the TEV in the context of water resources.[2]

3.2 Nonmarket Valuation Techniques

The development of nonmarket valuation techniques allows us to quantify various components of the TEV of water. Revealed preference techniques are employed to estimate use values, while stated preference techniques are appropriate for estimating both use and non-use values.

3.2.1 Revealed Preference Techniques

In this section, we introduce the two basic revealed preference techniques, widely used in environmental economics, for revealing the values individuals assign to an environmental asset: the hedonic pricing method and the travel cost method.

[2] Please see also National Research Council [5] book on groundwater valuation (Table 1.3, p. 20) for a taxonomy of groundwater values in particular. For example, according to this taxonomy, groundwater use values are divided into extractive (municipal, industrial and agricultural) values and in situ (ecological, buffer, subsidence avoidance, recreational and seawater intrusion values) use values.

Hedonic Pricing Method

This method uses the price variations of real estate market in order to estimate the value of a local environmental good or service. The main assumption behind this method is that people take into account local environmental characteristics when deciding to buy a property; therefore, the quality of the surrounding environment (such as air, water and noise pollution levels) will be reflected in the prices of real estate property. For example, Mahan et al. [6], based on a dataset of 14,000 home sales in Portland, found that proximity to wetlands had a positive effect on property values (a decrease in the distance to the nearest wetland by 1,000 feet caused property values to go up by $436). Likewise, in the context of groundwater, land rent and property prices can be used as shadow prices, i.e. as implicit values for estimating the value of water's quantity and quality. Torell et al. [7], for example, compared sales of irrigated and nonirrigated pieces of land in the southern High Plains (an area within various central US states such as Texas, Oklahoma and Kansas) and found that the value of groundwater was an important part of transaction prices for irrigated farmland (comprising from 30% to 60% of the farm sale price across the various states). King and Sinden [8] valued soil erosion and related effects on groundwater in New South Wales, Australia, and concluded that the market seemed to be working to conserve the soil in the examined region.

Travel Cost Method

This method is commonly used for estimating people's willingness to pay for visiting various ecosystem areas and natural landscapes for recreational activities. The basic assumption behind this method is that the value of the environmental amenity will be reflected on the time and travel cost that a person is willing to incur in order to access the site. The results of this method are used to determine changes in the access cost of a recreational site or to assess policy interventions with a view to improving environmental conditions. Bowker et al. [9], for example, employed the TCM in the Chattooga and Nantahala rivers in the USA and derived a value for guided white water rafting between $89 and $286 per visitor per trip. Wilson and Carpenter [10] estimated WTP for water quality changes in lakes, rivers, wetlands and streams in the USA (their estimates were $6 per trip to avoid further degradation in the considered 13 sites, $13 per trip to improve water quality boatable state to fishable state and $51 per trip to improve water quality from boatable to swimmable state).

3.2.2 Stated Preference Techniques

In contrast to revealed preference methods, capturing only use values, stated preference techniques are appropriate for measuring both use and non-use values from ecosystem services. Capturing and monetising the value of ecosystem services

Table 3 Scenario design criteria and contingent valuation measurement outcomes

Is the scenario...	If not, respondent will...	Measurement consequence
Theoretically accurate?	Value wrong things (theoretical misspecification)	Measure wrong thing
Policy relevant?	Value wrong things (policy misspecification)	Measure wrong thing
Understandable by respondent as intended?	Value wrong things (conceptual misspecification)	Measure wrong thing
Plausible to the respondent?	Substitute another condition, or not take seriously	Measure wrong thing. Unreliable, bias susceptible don't know or protest zero
Meaningful to the respondent?	Not take seriously	Unreliable, bias susceptible don't know or protest zero

Source: Mitchell and Carson [12], p. 190

may increase the efficiency of policy interventions, leading to an increase in environmental sustainability and net benefits for society [11]. This section reviews the two most popular methods of this type: the contingent valuation method and choice experiments.

Contingent Valuation Method

This method aims at eliciting people's willingness to pay (WTP) for *positive* changes in the quantity or quality of an environmental resource or their willingness to accept (WTA) compensation for *negative* changes in the status of the resource. It is a survey-based approach in which participants are asked to state their preference on a *hypothetical* scenario explained in the study. Therefore, the construction and implementation of the survey is a major challenge: particular care is required for the wording of the questionnaire and the administration of the survey so as to minimise bias. Table 3 summarises the basic criteria for a good scenario. Pate and Loomis [13] have provided a water-related application of CVM, in which households were willing to pay for the adoption of an improvement programme in a wetland in California. Hite et al. [14] also employed a CVM to assess public willingness to pay for reductions in agricultural nonpoint pollution and concluded that significant public support existed towards a policy providing farmers with precision application equipment to reduce nutrient runoff.

Choice Experiment Method

The choice experiment method (CEM) is a relatively new addition to the pool of stated preference techniques, having its theoretical foundations in Lancaster's [15] theory of value. The latter suggests that individuals derive satisfaction not by the consumption of a certain good itself but from its various attributes. Therefore, in choice experiments, a bundle of environmental goods is presented to respondents with various attributes or characteristics (price is usually one of the main attributes). Due to its experimental nature, the CEM enables researchers to evaluate attributes at various levels (e.g. high, medium or low status of water quality) and identify trade-offs that respondents have among the attributes. Each set of choices is then associated with a certain level of utility. Willis et al. [16] examined consumers' trade-offs between water supply security and river flows/biodiversity in local wetlands in Sussex, UK. Their findings suggest that though consumers assigned an insignificant value on increasing water supply, they had a positive value for a unit increase in the conservation of wetland habitats and river flows.

As a summary of this section, Table 4 presents the advantages and disadvantages of the main economic valuation methods. Also, it is worth noting that herein we have mentioned briefly only a couple of applications of revealed and stated preference techniques; nevertheless, the literature is vast (e.g. regarding the estimation of groundwater benefits, we refer the interested reader to Work Package 6-Genesis Project[3] for a thorough discussion of a large number of valuation studies, undertaken worldwide).

3.2.3 Laboratory Experiments

Laboratory experiments investigate preferences under a 'real setting' situation, fully controlled in a laboratory [19]. Real economic incentives are provided to the participants in order to reveal their WTP for a certain public or private good. Table 5 contains a brief description of some basic incentive-compatible mechanisms. For example, in the second-price sealed-bid Vickrey auction [21], participants submit sealed bids and the good is acquired by the participant who provides the highest bid, but at a price equal to the value of the second-highest bid. Several conditions may affect the quality of the performed experiments, such as the participants' unfamiliarity with the elicitation mechanisms, their tendency to use numbers (presented to them) as anchor values for their WTP, the presence of researchers scrutinising participants' behaviour and the use of a non-representative sample [20].

[3] Work Package 6 'Application of valuation techniques to assess the benefits of groundwater quantity-quality improvements' of the Genesis Project (Groundwater and Dependent Ecosystems: New Scientific and Technological Basis for Assessing Climate Change and Land-use Impacts on Groundwater). Genesis Project is available at: www.thegenesisproject.eu.

Table 4 Advantages and disadvantages of economic valuation methods[a]

Hedonic pricing method	*Advantages*
	Based on observable and readily available data from actual behaviour and choices
	Disadvantages
	Difficulty in detecting small effects of environmental quality factors on property prices Connection between implicit prices and value measures is technically complex and sometimes empirically unobtainable Ex post valuation (i.e. conducted after the change in environmental quality or quantity has occurred) Does not measure non-use values
Travel cost method	*Advantages*
	Based on observable data from actual behaviour and choices Relatively inexpensive
	Disadvantages
	Need for easily observable behaviour Limited to in situ resource use situations including travel Limited to assessment of the current situation Possible sample selection problems Ex post valuation Does not measure non-use values
Production function approach	*Advantages*
	Based on observable data from firms using water as an input Firmly grounded in microeconomic theory Relatively inexpensive
	Disadvantages
	Understates WTP Ex post valuation Does not measure non-use values Omits the disutility associated with illness
Contingent valuation	*Advantages*
	It can be used to measure the value of anything without need for observable behaviour (data) It can measure non-use values Technique is not generally difficult to understand Enables ex ante and ex post valuation
	Disadvantages
	Subject to various biases (e.g. interviewing bias, starting point bias, nonresponse bias, strategic bias, yea-saying bias, insensitivity to scope or embedding bias, payment vehicle bias, information bias, hypothetical bias) Expensive due to the need for thorough survey development and pre-testing Controversial for non-use value applications
Choice experiment method	*Advantages*
	It can be used to measure the value of any environmental resource without the need for observable behaviour (data), as well as the values of their multiple attributes It can measure non-use values

(continued)

Table 4 (continued)

	Eliminates several biases of CVM Enables ex ante and ex post valuation
	Disadvantages
	Technique can be difficult to understand Expensive due to the need for thorough survey development and pre-testing Controversial for non-use value applications

Source: Commission on Geosciences and Environment and Resources (CGER) [17], cited in Birol et al. [4], p. 114

[a]When time and budget constraints do not allow for the employment of an original valuation study, the benefit transfer method can be applied, i.e. economic estimations can be transferred from one study site to another with similar location characteristics. More details about this method can be found in Koundouri et al. [18]

Table 5 Incentive-compatible mechanisms

Elicitation mechanism	Participant procedure	Market price	Rule	# of winners
English auction	Sequentially offer ascending bids	Last offered bid	Highest bidder pays market price	1
2nd price auction	Simultaneously submit sealed bids	Second highest bid	Highest bidder pays market price	1
*N*th- price auction	Simultaneously submit sealed bids	*N*th highest bid	*n*-1 highest bidders pay market price	*n*-1
Random *N*th- price auction	Simultaneously submit sealed bids	Randomly drawn *N*th highest bid	*n*-1 highest bidders pay market price	*n*-1
Becker-DeGroot-Marschak	Simultaneously submit sealed bids	Randomly drawn price	Participant pays market price if bid exceeds market price	Individually determined
Real choice	Choose alternatives in multiple scenarios	Randomly drawn binding scenario	Everybody pays market price	All participants
Incentive-compatible conjoint ranking mechanism	Rank alternatives in multiple scenarios	Randomly drawn binding scenario	Everybody pays market price	All participants
Open-ended choice experiment	Simultaneously submit quantities	Randomly drawn price	Everybody pays market price for submitted quantities	All participants
Multiple price list	Accept/reject stated prices	Randomly drawn price	Participants pay market price if it is accepted	Individually determined
Real dichotomous choice experiment	Accept/reject	Given price	Participants pay market price if it is accepted	Individually determined
Quantity trade-off experiment	Accept/reject	No price	Participants complete trade if it is accepted	Individually determined

Source: Alfnes and Rickertsen [20], p. 219

3.3 *Integrated Hydro-Economic Models for Optimal Water Management*

In the previous section, we provided an overview of some common valuation techniques with regard to the calculation of various components of water's TEV. Now, we turn to hydro-economic models as tools for estimating water's economic value and suggesting strategies leading to an optimal water allocation.[4]

Integrated hydro-economic models are mathematical models combining hydrologic, engineering, environmental and economic aspects of water resource systems at a regional level [22]. They are used in order to suggest ways for more efficient and transparent use of water, given the existence of scarcity. The main assumption behind hydro-economic models is that demand for water may change subject to dynamic changes in water quantity and the type of use. Due to the various conditions that affect water availability (such as location and hydrologic conditions), more than one demand curves may be used [22].

Although hydro-economic models are driven by various institutional and socio-economic factors, the key focus is on the water system and its effect on one or more economic sectors [23]. Figure 1 depicts the disciplinary dimensions behind integrated hydro-economic models, and Table 6 provides a brief description of various types of hydro-economic models with their associated advantages and disadvantages.

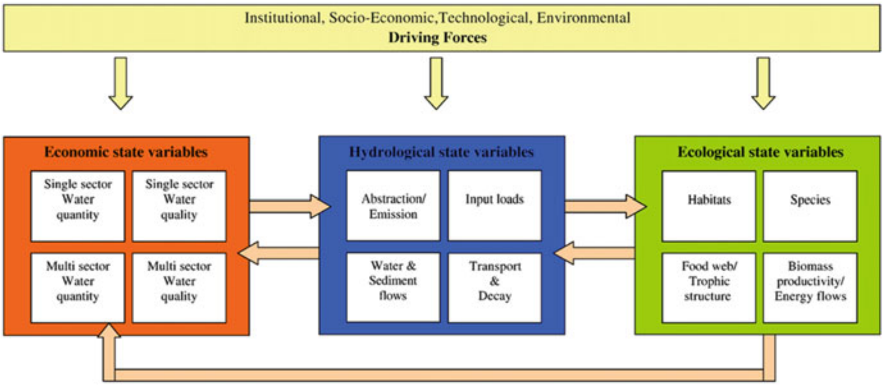


Fig. 1 Disciplinary dimensions underlying integrated hydro-economic modelling. *Source*: Brouwer and Hofkes [23], p. 17

[4] Apart from nonmarket valuation techniques and hydrological models, linear programming and various other econometric modelling approaches can be used for estimating the economic value of water. For example, programming models can be used for estimating the water quantity which maximises farmers' private profits through computer simulations (in cases where there is no data on a wide range of prices). These techniques are, nevertheless, beyond the scope of this review chapter.

Table 6 Some design choices, options and implications for building hydro-economic models

Simulation/ optimisation	
Simulation	
Summary	Time-marching, rule-based algorithms; answers question: 'what if?'
Advantages	Conceptually simple; existing simulation models can be used, reproduces complexity and rules of real systems
Disadvantages	Model only investigates simulated scenarios, requires trial and error to search for the best solution over wide feasibility region
Optimisation	
Summary	Maximises/minimises an objective subject to constraints[a]; answers question: 'what is best?'
Advantages	Optimal solutions can recommend system improvements; reveals what areas of decision space promising for detailed simulation
Disadvantages	Economic objectives require economic valuation of water uses; ideal solutions often assume perfect knowledge, central planning or complete institutional flexibility
Representing time	
Deterministic time series	Model inputs and decision variables are time series, historical or synthetically generated
Summary	Conceptually simple: easy to compare with time series of historical data or simulated results
Advantages	Inputs may not represent future conditions; limited representation of hydrologic uncertainty (system performance obtained just for a single sequence of events)
Disadvantages	
Stochastic and multistage stochastic	
Summary	Probability distributions of model parameters or inputs; use of multiple input sequences ('Monte Carlo' when equiprobable sequences or 'ensemble approach' if weighted)
Advantages	Accounts for stochasticity inherent in real systems
Disadvantages	Probability distributions must be estimated and synthetic time series generated; presentation of results more difficult; difficulties reproducing persistence (Hurst phenomenon) and non-stationarity of time series
Dynamic optimisation	
Summary	Inter-temporal substitution represented
Advantages	Considers the time-varying aspect of value; helps address sustainability issues
Disadvantages	Requires optimal control or dynamic programming
Submodel integration	
Modular	
Summary	Components of final model developed and run separately
Advantages	Easier to develop, calibrate and solve individual models
Disadvantages	Each model must be updated and run separately; difficult to connect models with different scales
Holistic	
Summary	All components housed in a single model

(continued)

Table 6 (continued)

Simulation/ optimisation	
Advantages	Easier to represent causal relationships and interdependencies and perform scenario analyses
Disadvantages	Must solve all models at once; increased complexity of holistic model; requires simpler model components

Source: Harou et al. [22], p. 632

[a]If optimised time horizon is a single time period, the model can be considered a simulation model that uses an optimisation computational engine

4 Rapid Assessment of the River Basin Districts in Greece

In this section, we provide a brief description of the socio-economic and water status of Greece's river basin districts.

Greece occupies a total area of 131,957 km^2 and consists of 14 river basin districts. Table 7 summarises information about the population, area and water uses in each RBD. Greek authorities have undertaken management plans in each RBD to characterise the ecological and chemical status of all water bodies (e.g. rivers, lakes, coastal areas, etc.). In some districts, authorities have aggregated across waterbodies to determine the overall quality status for each basin, while in others a characterisation is made separately for each type of waterbody (readers may want to consult each district's management plan for more information on the chemical and ecological parameters).[5]

5 Review of Representative Valuation Case Studies from Greece

In this section, we provide some representative examples of water-related valuation studies in Greece. All these studies have been developed, during the last decade, by the RESEES/ICRE8 team.[6] We would like to point out that in no sense is this current section meant to provide an exhaustive list of economic valuation methods in the entire country; our goal is to present a representative sample.

[5] Management Plans (in Greek) are available at http://wfd.ypeka.gr (last accessed 12/02/2015).

[6] The *International Centre for Research on the Environment and the Economy* (ICRE8) is the outcome of the evolution of the *Research Team on Socio-Economic and Environmental Sustainability* (ReSEES). More details about the team's research can be found at: http://www.icre8.eu/.

Table 7 Economic analysis of the most important water uses in each RBD

RBD	Population (2001)	Area (km^2)	Demand for supply (hm^3/year)	Demand for irrigation (hm^3/year)	Demand for industry (hm^3/year)	Demand for livestock (hm^3/year)
West Peloponnesus	331,180	7,235	35	180	16.4	2.8
North Peloponnesus	615,288	7,397	69.7	416	8.3	6.5
East Peloponnesus	288,285	8,442	31.7	330	7.1	4.6
West Sterea	312,516	10,199	44	340	0.39	7.84
Epirus	464,093	9,980	54	303	4	10
Attica	3,737,959	3,186	414.7	68.5	20.8	1.6
East Sterea	577,955	12,291	49.6	796	29.2	7.5
Thessaly	750,445	13,142	83	1 211	17	13
West Macedonia	596,891	13,624	140	938	83	95
Central Macedonia	1,362,190	10,146	7.77	463	0.26	Trivial
East Macedonia	412,732	7,320	47.7	816.3	16.2	5.8
Thrace	404,182	11,243	47.6	792.1	14.7	7.1
Crete	601,131	8,335	42.33	320	4.1	
Aegean Islands	508,807	9,103	37.19	80.20	1.24	

Source: Data in this table (except the last two rows) were collected by different studies in Greek ('Ολοκληρωμένα Σχέδια Διαχείρισης των Λεκάνων Απορροής της Χώρας, 2013') available at http://wfd.ypeka.gr (last accessed 12/02/2015). For the last two rows, studies were not yet available on-line, and data were taken by Koundouri et al. [24], p.13

5.1 *Production Function Approach (Duration Analysis): Crete*

Genius et al. [25] developed a model to investigate the potential effect of information transmission on the adoption and diffusion of modern irrigation technology in agriculture. Information transmission was considered through two main sources: extension agents and social learning (i.e. interaction with peer farmers and learning by doing). The model was tested empirically through a dataset of 265 olive growers located in the island of Crete. The dataset included information about the year in which farmers adopted a new irrigation technology (such as drip or sprinklers) and about key farming-operation variables, such as production patterns, gross revenues, input use, water cost and the farmers' socio-demographic characteristics. According to the available data, none of the farmers had adopted a new technology before 1994, whereas 64.9% (172) of farmers had adopted a drip technology between 1994 and 2004. The mean adoption time for the sample was 4.68 years.

Using duration analysis, the authors found that both extension services and social interaction with peer farmers had been essential for the adoption and diffusion of new technology. Moreover, the two aforementioned channels were found to be complementary. Other variables affecting the decision to adopt the new technology were water and crop prices (water prices being positively while crop prices negatively associated with the adoption time), risk attitudes (risk-avert farmers being more likely to adopt the new irrigation technology), climatic conditions (adverse conditions, as in the case of Crete, which is characterised by a semiarid climate, were positively associated the adoption time) and some socio-demographic characteristics (e.g. the adoption time decreased with farmers' age up to 60 years but thereafter increased, thus highlighting the combined effect of planning horizon and farming experience).

5.2 Choice Experiment, Lab Experiment, Contingent Valuation: The Asopos River Basin

The Asopos river basin runs in the Eastern RBD of Greece, approximately 60 km north of Athens, and serves a population of approximately 70,575 citizens. The Asopos area constitutes the largest industrial region in Greece. The river and groundwater of the basin have been subject to long-term (since the 1970s) industrial and agricultural pollution. Agriculture plays an important role on water quality due to nitrate runoff from the excessive use of fertilisers, while industries create major environmental problems due to the lack of a holistic plan for the treatment of the produced industrial (liquid, solid and air) wastes. As a result, Asopos has been characterised as one of the most polluted rivers in Greece, having an impact not only on the areas that it runs through but also on the coastal area which it flows into. Asopos's serious environmental degradation, coupled with two different sub-population groups with regard to socio-economic characteristics (rural local residents vs. vacation urban residents from Athens), makes this case study particularly interesting.

For this purpose, Koundouri et al. [26] conducted a choice experiment in order to calculate the WTP of the two sub-population groups (Asopos and Athens residents) for improvements in environmental conditions. Following common practice, the CE survey included the following steps: (1) selection of attributes, (2) definition of attribute levels, (3) choice of experimental design in order to allocate alternative scenarios to choice tasks and (4) elicitation of preferences, based on respondents' ranking of available scenarios in each choice task. Table 8 presents the main attributes and the corresponding levels in various policy plans presented to the respondents. The results (Table 9) show that respondents from both sub-populations had significant marginal WTP for alternative policy scenarios improving local environmental conditions.

Moreover, a lab experiment [27] was conducted to examine the impact of environmental degradation on health and the cost from consuming products produced in an area with poor water conditions. A sample of 61 consumers were

Table 8 Attributes and levels

Attribute	Status quo (option A)	Some policy action
Environmental conditions	Bad	Moderate or good
Impact on local economy	Negative today	Improved by 2015 or positive by 2027
Human health	Water not suitable for drinking, cooking and irrigation	Water suitable for all uses (drinking, cooking and irrigation) or water suitable for some uses (drinking and cooking)
Cost in Euro (tri-monthly water bill per household for the next 15 years)	0	2, 4, 6, 8 or 12

Source: Koundouri et al. [26], p. 105

Table 9 Marginal WTP for the two sub-populations (all respondents)

Attribute level	Marginal WTP (Athens)	Marginal WTP (Asopos)
Status quo policy option	7.28***	8.31***
Environmental conditions: moderate	10.07***	9.59***
Environmental conditions: good	2.41***	0.47
Local economy improved by 2015	4.03***	1.70***
Local economy positive by 2027	−1.78***	−1.13***
Water for some uses	5.68***	7.29***
Water for all uses	6.27***	5.16***

Source: Koundouri et al. [26]
Note: Marginal WTP for status quo becomes insignificant when serial non-participants are excluded, i.e. those that are not satisfied by none of the alternative policy scenarios
*** is 99% significance level

recruited in Athens to participate in a 4th price Vickrey auction performed in the lab: after a brief training on the lab experiment process, participants were asked to bid in order to exchange a kilo of potatoes produced in the Asopos area with a kilo of potatoes produced in a region with good ecological status. Bids were modelled as a function of respondents' socio-economic characteristics, initial monetary endowment, risk perceptions and potato consumption habits. Estimates were obtained through a random effects regression model. The results suggest that participants were willing to pay a price premium in order to exchange the Asopos potatoes with potatoes from a less polluted region (the mean upgrade bid from lower to upper quality potatoes was found to be €0.60 euro per kilo). Moreover, participants were willing to pay in order to reduce their potential health risk even when they were informed that there would be no available data for assessing risks of consumption to human health.

Also, Tentes and Damigos [28] and Tentes et al. [29] have conducted two economic valuation studies, a contingent valuation and a choice experiment, respectively, in the Asopos area with a view to estimating environmental damage to groundwater. WTP estimates from both studies fall into the same range of values. Different household profiles showed different willingness to pay, depending on attitudes against the environmental damage, population age and place of residence

[28]. Households were willing to pay almost 160 €/month for in situ remediation measures at certain areas which suffer most, in order to serve all groundwater uses [29].

5.3 Choice Experiment: Cheimaditida Wetland

The wetland of Cheimaditida, located 40 kilometres southeast of Florina in the northwest part of Greece, covers an area of 168 km^2 and contains one of the last remaining freshwater lakes in Greece. Rich fauna, flora and habitat diversity can be met in the wetland. However, the economic activity in the area (mainly agriculture, forestry and fishing) has caused negative effects on the water quantity and quality and in turn on the wetland's rich biodiversity.

Birol et al. [30] conducted a choice experiment in order to estimate the value of the benefits derived by the wetland. Face-to-face interviews were employed in eight towns and two cities (Athens and Thessaloniki) representing a continuum of distances from the wetland, as well as urban and rural populations. Table 10 summarises the main attributes and their various levels presented to the study participants: two ecological (biodiversity and open-water surface area), two socio-economic (research/education and retraining of farmers) and one monetary attribute were selected. Different combinations of these attributes yielded the following management scenarios: (1) current scenario ('status quo'), i.e. low biodiversity, low water surface area, low research and educational opportunities and no farmers' retraining; (2) scenario 1 (low impact), i.e. low biodiversity, higher levels of open-water surface area, low research and educational opportunities and retraining of 30 farmers; (3) scenario 2 (medium impact), i.e. high level of biodiversity, low open-water surface area, high research and educational opportunities and retraining of 75 farmers; and (4) scenario 3 (high impact), high level of biodiversity, high open-water surface area, high research and educational opportunities and retraining of 150 local farmers. The payment vehicle was a one-off tax payment for the year 2006–2007 deposited to the 'Cheimaditida Wetland Management Fund', controlled by a credible and independent body. The collected dataset, besides responses on the various management plan scenarios, included socio-economic characteristics and the participants' attitudes towards the environment.

The econometric analysis (four basic conditional logit models) revealed that respondents were willing to pay in order to promote all attributes of the choice experiment: WTP varied between €15.10 and €17.8 for improvements in biodiversity, €7.25 and €11.02 for improvements in open-water surface area, €8.69 and €10.79 for education and research opportunities and €0.075 and €0.195 for farmers' retraining. Taking into account the existence of potential heterogeneity among respondents' preferences, people with higher levels of education, income and environmental consciousness appear to prefer management scenarios with higher levels of ecological and socio-economic attributes. Also, the compensating surplus increased when moving from the status quo to one of the alternative scenarios for

Table 10 Wetland management attributes and levels used in the CE

Attribute	Definition	Management level
Biodiversity	The number of different species of plants, animals, their population levels, the number of different habitats and their size	Low: deterioration from current levels High: a 10% increase in population and size of habitats
Open-water surface area	The surface area of the lake that remains uncovered by reed beds	Low: decrease from the current open-water surface area of 20% High: increase open-water surface area to 60%
Research and education	The educational, research and cultural information that may be derived from the existence of the wetland, including visits by scientists, students and school children to learn about ecology and nature	Low: deterioration from the current levels of opportunities High: improve the level of educational and research opportunities by providing better facilities
Retraining of farmers	Retraining of local farmers in environmentally friendly employment such as eco-tourism and arid-crop production	Number of farmers retrained in environmentally friendly employment: 30, 50, 75, 150
Payment	A one-off payment to go to the 'Cheimaditida Wetland Management Fund'	4 payment levels from the pilot CV: 3, 10, 40, 80

Source: Birol et al. [30], p. 147

the management of the wetland. Subject to various model specifications, the WTP ranged between €58.2 and €107.59 for the low-impact scenario, €80.11 and €116.49 for the medium-impact scenario and €102.69 and €134.46 for the high-impact scenario. Finally, a cost-benefit analysis was employed to calculate the net benefits generated by each of the three aforementioned scenarios. The estimated aggregate net benefits were €335.351.463, €357.421.769 and €412.825.286 for the low-, medium- and high-impact management scenarios respectively, indicating that social welfare maximises with the high-impact scenario.

6 Conclusion and Policy Implications

Economic analysis needs to be integrated with other field expertise (climate change, hydrology, geology, engineering, sociology, etc.) and be considered along the management and decision-making process. The main purpose of this chapter was to discuss how economic analysis can assist in achieving the targets of the EC WFD in terms of designing efficient, socially equitable and environmentally sustainable water management policies. In summary, integrating economic analysis in an interdisciplinary management effort towards implementing the WFD entails identifying the uses of the RB services for different sectors of the local economy and estimating their monetary value. Estimating such values is an important prerequisite for the design of appropriate policies, leading to sustainable management over time and space.

A variety of valuation methods, primarily revealed and stated preferences methods, is available to economists in order to identify and quantify economic values, use and non-use, arising from the various aquatic resources and ecosystems.

A brief description of water-related empirical studies conducted in Greece, during the last decade, reveals that various socio-demographic characteristics and different stakeholders' interests affect perceptions and willingness to pay for a certain environmental good or policy intervention. Comparing the benefits, for all water users, yielded by the implementation of various water-related management scenarios, to the associated costs, allows us to identify a socially efficient wetland management strategy. In other words, CBA may be helpful in the avoidance of policy interventions with disproportionate costs to member states.

In order to achieve a socially efficient policy and WFD compliance 'good status', it is essential to employ economic instruments that allow us to impose the payment of the total economic values of RB on the users of these services. For instance, financial penalties could be imposed on pollution dischargers; alternatively, subsidies could be paid directly by the central or local government to the provider of water services in the form of investment subsidies (capital subsidies, lowering fixed costs). In this way, users would internalise (in their individual decisions) the social value of the resource and would thus be incentivised to use it in a sustainable manner. The effects of such policy interventions on different social groups could vary but could also be smoothed out with a redistributional instrument subject to explicit government priorities, approved by democratic processes.

References

1. Commission of European Communities (CEC) (2000) Directive 2000/60/EC of the European Parliament and of the Council establishing a framework for Community action in the field of water policy. Off J Eur Communities. http://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32000L0060&from=EN. Accessed 12 Feb 2015
2. Koundouri P, Scarpa R, Stithou M (2014) A choice experiment for the estimation of the economic value of the river ecosystem: Management policies for sustaining NATURA (2000) species and the coastal environment. In: Koundouri P, Papandreou N (eds) Water resources management sustaining socio-economic welfare: the implementation of the European water framework directive in Asopos river basin in Greece. Global Issues in Water Policy. Springer, Dordrecht-Heidelberg-New York-London
3. European Communities (2003) Common Implementation Strategy for the Water Framework Directive (2000/60/EC). Guidance document No. 1, economics and the environment – the implementation challenge of the Water Framework Directive. Office for Official Publications of the European Communities, Luxembourg. http://www.enorasis.eu/uploads/files/Water%20Governance/9.Guidancedoc01_Economics_WATECO_WG2-6.pdf. Accessed 12 Feb 2015
4. Birol E, Karousakis K, Koundouri P (2006) Using economic valuation techniques to inform water resources management: A survey and critical appraisal of available techniques and an application. Sci Total Environ 365:105–122

5. Committee on Valuing Ground Water, National Research Council (1997) Valuing ground water: economic concepts and approaches. The National Academies Press, Washington, DC
6. Mahan B, Polasky S, Adams R (2000) Valuing urban wetlands: A property price approach. Land Econ 76:100–113
7. Torell A, Libbin J, Miller M (1990) The market value of water in the Ogallala aquifer. Land Econ 66:163–175
8. King D, Sinden A (1988) Influence of soil conservation on farm land values. Land Econ 64 (3):242–255
9. Bowker JM, English D, Donovan J (1996) Toward a value for guided rafting on southern rivers. J Agric Appl Econ 28:423–432
10. Wilson M, Carpenter S (1999) Economic valuation of freshwater ecosystem services in the United States: 1971–1997. Ecol Appl 9:772–783
11. Department for Environment Food and Rural Affairs (DEFRA) (2007) An introductory guide to valuing ecosystem services. DEFRA, London. https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/221092/pb12852-eco-valuing.pdf. Accessed 12 Feb 2015
12. Mitchell R, Carson R (1988) Evaluating the validity of contingent valuation studies. In: Peterson G, Driver B, Gregory R (eds) Amenity resource valuation: Integrating economics with other disciplines. Venture Publishing Inc, State College
13. Pate J, Loomis J (1997) The effect of distance on willingness to pay values: a case study of wetlands and salmon in California. Ecol Econ 20:199–207
14. Hite D, Hudson D, Intarapapong W (2002) Willingness to pay for water quality improvements: the case of precision application technology. J Agric Resour Econ 27(2):433–449
15. Lancaster K (1966) A new approach to consumer theory. J Polit Econ 84:132–157
16. Willis K, Mcmahon P, Garrod G, Powe N (2002) Water companies' services performance and environmental trade-offs. J Environ Plann 45(3):363–379
17. Commission on Geosciences, Environment and Resources (CGER) (1997) Valuing ground water: Economic concepts and approaches. http://www.nap.edu/openbook.php?isbn=0309056403&page=75. Accessed 12 Feb 2015
18. Koundouri P, Papandreou N, Remoundou K, Kountouris Y (2014) A bird's eye view of the Greek Water Situation: The potential for the implementation of the EU WFD. In: Koundouri P, Papandreou N (eds) Water resources management sustaining socio-economic welfare: the implementation of the European water framework directive in Asopos river basin in Greece, Global Issues in Water Policy. Springer, Dordrecht-Heidelberg-New York-London
19. Murphy J, Stevens T (2004) Contingent valuation, hypothetical bias, and experimental economics. Agric Resour Econ Rev 33(2):182–192
20. Alfnes F, Rickertsen K (2011) Non-market valuation: experimental methods. In: Lusk JL, Roosen J, Shogren JF (eds) The Oxford handbook of the economics of food consumption and policy. Oxford University Press Inc, New York
21. Vickrey W (1961) Counterspeculation, auctions, and competitive sealed tenders. J Finance 16 (1):8–37
22. Harou J, Pulido-Velasquez M, Rosenberg D, Medellín -Azuara, J, Lund, R, Howitt R (2009) Hydro-economic models: concepts, design, applications, and future prospects. J Hydrol 375 (3–4):627–643
23. Brouwer R, Hofkes M (2008) Integrated hydro-economic modelling: approaches, key issues and future research directions. Ecol Econ 66:16–22
24. Koundouri P, Remoundou K, Kountouris Y (2010) A Note on the Implementation of the Economics of the EU Water Framework Directive under Data Limitations: a rapid appraisal approach. Department of International and European Economic Studies, Athens University of Economics and Business, Working Paper 1008. http://wpa.deos.aueb.gr/wpa_paper_list.php
25. Genius M, Koundouri P, Nauges C, Tsouvelekas V (2013) Information transmission in irrigation technology adoption and diffusion: social learning, extension services and spatial effects. Am J Agric Econ 96(1):328–344

26. Koundouri P, Papandreou NA (eds) (2014) Water resources management sustaining socio-economic welfare the implementation of the European water framework directive in Asopos River Basin in Greece. Global issues in water policy series, vol 7. ISBN:978-94-007-7635-7 (Print) 978-94-007-7636-4 (Online)
27. Drichoutis A, Koundouri P, Remoundou K (2014) A laboratory experiment for the estimation of health risks: policy recommendations. In: Koundouri P, Papandreou N (eds) Water resources management sustaining socio-economic welfare: the implementation of the European water framework directive in Asopos river basin in Greece, Global Issues in Water Policy. Springer, Dordrecht-Heidelberg-New York-London
28. Tentes G, Damigos D (2012) The lost value of groundwater: the case of Asopos River Basin in Central Greece. Water Resour Manag 26(1):147–164
29. Tentes G, Tsiotsia A., Damigos, D (2013) Environmental cost of groundwater pollution: a choice experiment in Asopos River basin. In: Proceedings of the 1st EWaS-MED international conference "improving efficiency of water systems in a changing natural and financial environment", 11–13 April. Thessaloniki, Greece
30. Birol E, Karousakis K, Koundouri P (2006) Using a choice experiment to account for preference heterogeneity in wetland attributes: the case of Cheimaditida wetland in Greece. Ecol Econ 60:145–156

Environmental Impacts of Large-Scale Hydropower Projects and Applied Ecohydrology Solutions for Watershed Restoration: The Case of Nestos River, Northern Greece

Georgios Sylaios and Nikolaos Kamidis

Abstract The impoundment and abstraction of freshwater in river systems for the purposes of power generation and agricultural irrigation has provided huge economic benefits at global scale over the last 50 years. However, the environmental and social costs induced by large dams have been poorly accounted for in economic terms. Although the construction of large dams reduces the threat of devastation from extreme floods, significant watershed changes arise as a result of reservoir filling, river flow blockage, river flow storage, and flow regulation. Almost 15 years after the construction and operation of the two large hydropower dams (Thissavros and Platanovrisi), and approximately 50 years after the operation of the irrigation dam at Toxotes, the environmental consequences at the downstream part, the deltaic and the coastal zone of Nestos River are evident. The application of specific ecohydrological concepts aims to mitigate these observed environmental effects. Ecohydrology is an innovative and exciting topic, considered as the science of interplay between biota and hydrology in an ecosystem. Riparian and coastal ecohydrology concepts, together with the concept of "Green Hydropower", may be adopted as a tool for assessing the effectiveness of the measures adopted by the hydropower companies aiming at improving the environmental functioning of such systems.

G. Sylaios (✉)
Laboratory of Ecological Engineering and Technology, Department of Environmental Engineering, Democritus University of Thrace, 67100 Xanthi, Greece
e-mail: gsylaios@env.duth.gr

N. Kamidis
ELGO-Demeter, Fisheries Research Institute, 64007 Nea Peramos, Kavala, Greece
e-mail: nikkami@inale.gr

N. Skoulikidis et al. (eds.), *The Rivers of Greece: Evolution, Current Status and Perspectives*, Hdb Env Chem (2018) 59: 379–402, DOI 10.1007/698_2017_473,

Keywords Damming, Ecohydrology, River flow blockage, River flow regulation, River restoration

Contents

1 Introduction

Natural flow in the majority of the world's rivers has been altered by humans, through dam construction, water diversion and abstraction to cover present and future water and energy needs [1]. As water and energy are indispensable for human sustenance, the resulted impoundment of freshwater resources for water supply, hydropower generation, and agricultural irrigation has provided huge economic benefits at a global scale over the last 50 years [2]. In parallel, the growing population and the rising level of economic activity increase human demand for water and water-related services. For example, the electric power demand is expected to rise by almost 55% until 2030, with countries like China and India presently covering about the half of this demand [3].

To cover these needs, the mean construction rate of large dams, worldwide, but excluding China, ranges from 160 to 320 large dams/year. Presently, there exist more than 45,000 dams operating in over 150 countries, with height over 15 m, leading to the impoundment of 8,400 km^3 of water stored in reservoirs, seven times higher than the water volume of 1,200 km^3 in natural rivers [4]. These dams generate approximately 19% of the world's electricity and irrigate almost 40% of the irrigated land worldwide [5]. Global hydropower generation is expected to rise by 1.7% per year between 2004 and 2030, depicting an overall increase of almost 60% since 2004, as global investment in large dams in 2001 exceeded $2 trillion [6].

On the other hand, the projected climatic change impacts associated with the decreasing trend in the precipitation patterns seem likely to induce water scarcity, leading to lower river flow levels, especially over the Mediterranean basin

[7]. Qualitative degradation of river water quality is also expected, resulted from the projected exponential rise in the water requirements for food production and the consequent increase in the utilization of fertilizers and pesticides.

All the above lead to the conclusion that humans will continue to exert significant pressure in the near future on rivers, and especially on dam-controlled rivers, thus their environmental impacts on the watershed ecosystem should comprehensively be revealed. Indeed, the environmental and social costs of large dams have been poorly accounted for in economic terms, so that the wider long-term cost/benefit analysis to determine the true profitability of these projects remains elusive [5]. It is presently apparent that river damming changes watershed biosphere, creating a completely different hydrologic and ecologic regime with more severe alterations at the downstream part of the watershed, affecting river deltas and the broader coastal zone and coastal ecosystem [8].

Over the years several investigators reviewed the impacts of river damming, with special reference to the downstream part of the watershed: Brismar [9] in a pioneer work presented extensively the cumulative environmental consequences related to large dam projects, by classifying them into impacts related to reservoir filling, river flow blockage, water storage, and flow regulation; Collier et al. [10] and Lin [11] discussed the downstream hydrological effects associated with water flow regulation due to river damming; Lerer and Scudder [12] and Tajziehchi et al. [13] examined among others the environmental, social, and health issues related to large dam projects, especially in less-developed countries.

The specific physical, chemical, and biological changes to the ecosystems of upstream backwaters, the reservoir body and surroundings, and the downstream river reaches, associated with river damming, are summarized as follows:

1. River water stage regulation and frequent short-term flooding incidents, as a result of energy-demand peaks [14, 15];
2. Dams smooth the hydrologic peaks and floods, while freshwater abstractions reduce the river flux reaching the coastal sea [16];
3. River water temperature regulation, with generally lower temperatures prevailing throughout the year [17];
4. Retention of nutrients in the reservoir, thus changing the stoichiometry of river water transported to the sea [18];
5. Changes in the downstream water quality which has effects on turbidity, dissolved gases, concentration of heavy metals and minerals [19];
6. Reduction of sediment loads at the deltaic and coastal zone areas, resulting in the alteration of coastal sedimentary budget and the occurrence of coastal erosion [20–23];
7. Stoichiometric change leading to biogeochemical cycling alterations for dissolved and particulate substances in coastal ecosystems [24];
8. Reduction of biodiversity due to the blocking of the movement of the organisms and because of environmental changes described above;
9. Changes in aquatic flora and fauna, with significant changes in river fish fauna, due to physico-chemical alterations [25, 26];

10. Primary production shift from the coastal sea to the lacustrine environment developed at dam's reservoir;
11. Impacts on the spawning and nursery grounds of riverine and coastal fish species, influencing coastal fisheries [27, 28].

From the above it occurs that dams affect (a) the hydrological system and river dynamics, (b) river physico-chemical cycles, (c) soil, surface, and groundwater quality and environmental toxicity, and (d) the watershed ecosystem. The scope of the present paper is to investigate the environmental changes taking place along the transboundary Nestos River (N. Greece), over the last 15 years since the construction and operation of two large hydropower dams (Thissavros and Platanovrisi), and to discuss a series of mitigation and restoration measures that have been (or could be) adopted, based on the modern principles of ecohydrology [29, 30] and "Green Hydropower" [31]. Impacts assessment is based on existing datasets obtained during previous studies and projects carried out either solely over the Greek part or covering the whole watershed, funded mostly by cross-border collaborating programs (e.g., Interreg III and IV) and environmental restoration initiatives (e.g., LIFE).

2 Site Description

2.1 Watershed Geology

Nestos/Mesta River is one of the 71 internationally shared river catchments of Europe [32], located in Eco-region 7 (Eastern Balkans), according to the Water Framework Directive 2000/60/EC (Fig. 1). Nestos/Mesta River valley is confined between the Pirin mountain in the West, the Rila mountain in the North, and the Rhodope massif in the East. The River drains 5,613 km^2, of which 2,770 km^2 (or 49.34% of the total basin) belong to Bulgaria, while 2,843 km^2 (or 50.66% of the total basin) is located in Greece [33]. In Bulgaria the river flows through a valley of granite, until it reaches the Rhodope mountain chain, near the Greek–Bulgarian borders, where a rugged mountainous terrain of Precambrian and early Paleozoic crystalline rocks prevails. Two glaciations seem to have taken place in the mountainous part of the basin, leading to the partial crushings of the granite peaks and the formation of beautiful alpine edges, moraine fields, and vertical cliffs. After the borders the river continues to flow on metamorphic rocks (gneisses, schists), until the karstified marbles of Paschalia area. The impressive "Nestos Gorges" are formed in this karstic zone between Stavroupoli and Toxotes, characterized by the presence of steep slopes and meanders. At the deltaic zone, a floodplain is formed covering 440 km^2 of thick tertiary fluvial sediments belonging to Prinos–Nestos formation [34].

Fig. 1 The Mesta/Nestos River transboundary watershed

2.2 *River Hydrology*

The Nestos/Mesta River catchment experiences the influence of the European continental climate over the entire river flow. The southern-most part of the river valley serves as a corridor for the Mediterranean climate [35]. In the Bulgarian part mean annual precipitation reached 810 mm/year, while in the Greek part 790 mm/year. Mesta River is supplied by 25 tributaries, 9 of them are considered as first order, as they transfer large water quantities. The biggest is Dospatska or Dospat River, with a catchment area of 680 km^2, which flows into Mesta in the Greek territory (Fig. 1). In the Greek part, Nestos River is supplied by 18 tributaries, with Arkoudorema, Diavolorema, and Musdarema being the most important, having a total catchment area of 1,350 km^2 [36].

Examination of the Nestos River mean monthly flow time-series at Temenos site, during the period 1966–2000, revealed that the mean pre-damming annual discharge was 39.7 m^3/s and the annual discharge 1,120 Mm^3, characterized by a strong seasonal and inter-decadal variability [33]. The maximum mean annual discharge was observed in year 1996 at 75.48 m^3/s (annual runoff 2,380 $\times$ 10^6 m^3), while a minimum mean annual discharge of 15 m^3/s (annual runoff 473.5 $\times$ 10^6 m^3) was recorded in 1993.

2.3 Effects of Climate Change and Human Intervention on River's Hydrology

Koutroumanidis et al. [37] analyzed the 1966–1996 water discharge time-series recorded at the border station Delta, upstream Thissavros and Platanovrisi dam reservoirs, in an attempt to identify structural change points, as a result of climatic and/or human interventions. They used the "fuzzy entropy" criterion, as a measure of disorder within the hydrologic time-series, and the RCUSUM statistic to identify the time-series phase change point. The conclusion that the Temenos series was inhomogeneous and that it could be divided into five parts (generations) was finally reached (Fig. 2).

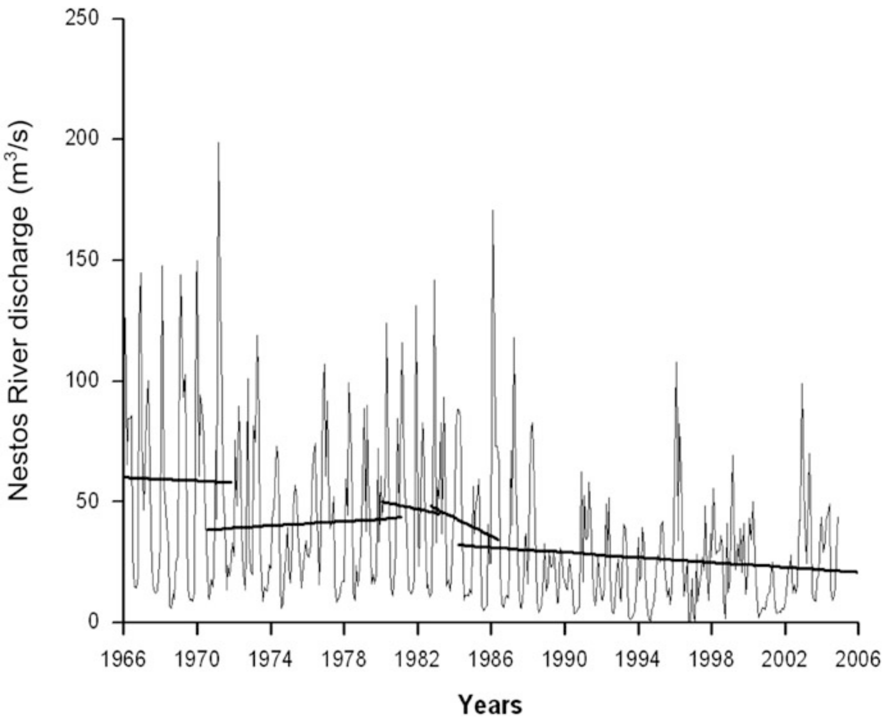


Fig. 2 Nestos River discharge time-series measured upstream of Thissavros Hydropower Reservoir, during 1996–2006 (source: Greek Public Power Corporation)

Generation 1 covers the period from January 1966 to May 1971, with a mean annual discharge of 36.4 m^3/s and a total annual runoff of 1,132 × 10^6 m^3/year. During Generation 2 (June 1971 to April 1980) a 21.8% reduction in the total annual runoff was observed, but the generation trend remains positive. Indeed, Nestos River discharge increases slightly during Generation 3 (May 1980 to December 1982) to the level of 29.6 m^3/s. However, a significant reduction in Nestos River flow was recorded during the period January 1983 to January 1986 (Generation 4) to 22.3 m^3/s, depicting a strongly negative trend. Finally, mean monthly discharge reached 17.7 m^3/s corresponding to a total annual surface runoff of only 552 × 10^6 m^3/year, almost half compared to the first runoff period (Table 1). Flow reduction in this fifth generation coincides well with the recorded trend in precipitation at the northern part of Greece, with a rate of 0.57 mm/winter/year [38]. Finally, the decomposition of the de-centered and de-trended above-defined generations into their periodic components revealed the presence of periods similar to those found in almost all world's river discharge and precipitation series [37].

All the above indicate that climatic factors and intensified human usage through damming, abstraction, and diversion decreased significantly the annual river flow crossing the Greek–Bulgarian borders, during the last 30 years. Such reduction is expected to exert in the future an important pressure at the dams' downstream river part, thus enhancing environmental impacts.

Table 1 Hydrologic characteristics of the five generations defined in the Nestos River discharge time-series

Generation	Time period	Mean annual discharge (m^3/s)	Total annual runoff (×10^6 m^3/year)	Mean daily runoff (×10^3 m^3/day)	Annual runoff change from generation 1 (%)
1	January 1966–May 1971	36.41	1,132	3,102.36	0.00
2	June 1971–April 1980	28.47	886	2,426.30	−21.79
3	May 1980–December 1982	29.59	920	2,521.34	−18.73
4	January 1983–January 1986	22.31	694	1,900.89	−38.73
5	February 1986–December 2004	17.76	552	1,513.50	−51.21

2.4 Present Nestos/Mesta Water Uses

The water of Mesta/Nestos River in Bulgaria is mainly used for irrigation, domestic use, fishery, tourism, waste disposal, and slightly for energy production. On the other hand, this water in Greece is mostly used for energy production, irrigation, domestic use, fishery, waste disposal, and tourism [39].

The water use for energy production in Bulgaria is limited. Three small hydropower stations exist in Mesta basin (Yakoruda, Razlog, and Toplika) producing in total 7.4 KWh. Extensive water use for energy production takes place in Greece. The Public Electricity Corporation constructed and operates since 1996 the multipurpose dams of Thissavros and Platanovrisi. Their characteristics are given in Table 2. On the Bulgarian drainage basin of Mesta, the dam of Despat River was constructed and operated well before the dams on the Hellenic side [40]. This dam diverts approximately 378×10^6 m^3 towards the Maritsa/Evros basin.

In terms of irrigation, four extensive irrigation networks have been developed along Mesta/Nestos River (two in Bulgaria and two in Greece), aiming to supply with water the adjacent cultivations. In Bulgaria, approximately 50×10^6 m^3 are abstracted from Mesta/Nestos river, to cover the irrigational needs of 150 km^2 cultivated land. In the Greek part the total arable land reaches 142 km^2, of which 131 km^2 are located at the deltaic area of the river and only 11 km^2 are situated at its central and northern part. Their total annual water demands exceed the level of 200 million m^3 [33].

In Bulgaria, in terms of domestic water use, approximately 5.2×10^6 m^3/year are abstracted to cover human consumption utilizing surface (56%) and groundwater resources (34%). Industrial activities abstract water of the order of 3.8×10^6 m^3/year, especially at the towns of Razlog and Belitsa. In the Greek part, the domestic water use of 5×10^6 m^3/year seems balanced by the kartsic springs supply at the southern part of Lekani marbles (springs of Paradeisos and Stratones).

Table 2 Descriptive data for Thissavros and Platanovrisi dams and reservoirs

	Thissavros dam	Platanovrisi dam
Mean width (m)	200	200
Mean depth (m)	40	15
Highest operational stage from MSL (m)	+380	+227.5
Lowest operational stage from MSL (m)	+320	+195
Volume at HOS (10^6 m^3)	705	73
Volume at LOS (10^6 m^3)	135	16
Reservoir area in HOS (km^2)	18	3
Maximum annual stage fluctuation (m)	30	2

3 Environmental Impacts Related to Nestos River Damming

3.1 Impacts Related to Reservoir Filling

After river damming two large reservoirs were developed: Thissavros Reservoir, approximately 32.8 km in length, and Platanovrisi Reservoir, about 11 km in length. These reservoirs inundated significant parts of mountainous forested land and altered the hydrologic, biogeochemical, and ecological conditions of the system. As pre-damming deforestation of inundated land was kept to minimal, the burial of forested land at the reservoirs' bottom enhanced the decomposition of organic matter, released nutrients into the water column, and promoted anaerobic conditions at the near-bottom zone. Figure 3 illustrates an indicative distribution of dissolved oxygen along Thissavros and Platanovrisi Reservoirs, during summer 2006 (after [41]). The most common pattern appearing in all monthly transects in both reservoirs is the increased DO-levels at the surface and the DO depletion near the bottom (at depths above 40 m). Thermal stratification in the water column during spring and summer obstructs vertical mixing and reduces DO bottom levels near hypoxia (3.1 mg/L at Thissavros and 2.6 mg/L at Platanovrisi). Although never measured, under such conditions the emission of methane gas and the production of hydrogen sulfate and methyl mercury at the reservoirs bottom seem possible.

The thermal, nutrients, and primary production cycle of Platanovrisi Reservoir was monitored throughout 2007, with 20-day repeatability [42]. In parallel, an one-dimensional (z-axis) numerical model was developed to simulate the reservoir's water column dynamics. Results showed that the stratification–destratification cycle strongly affects the vertical distribution of nutrients, as during the thermocline presence (approximately Julian day 100–300) the hypolimnetic levels of nitrates, nitrites, and total phosphorus were up to five times higher than those observed at the epilimnion. The reverse behavior was exhibited by the dissolved oxygen concentration, leading to anoxic values during the stratified period near the bottom (Fig. 4).

Further, the change from riverine to lake conditions affected directly the biodiversity of the ecosystem. Koutrakis et al. [43] reported that the fish fauna in Nestos River reservoirs is now dominated by limnophilic species, in their majority alien and translocated species, favoring the post-damming established lacustrine environment, while native species appear limited in richness and diversity.

3.2 Impacts Related to Flow Blockage

After dams' development, Nestos river inflows into Thissavros Reservoir at significantly lower speeds, allowed the deposition of bedload and suspended sediment at the entrance of the reservoir. Similarly, the entrapment and burial of solid wastes, transferred from Bulgaria after heavy rainfall incidents, at Thissavros entrance, has

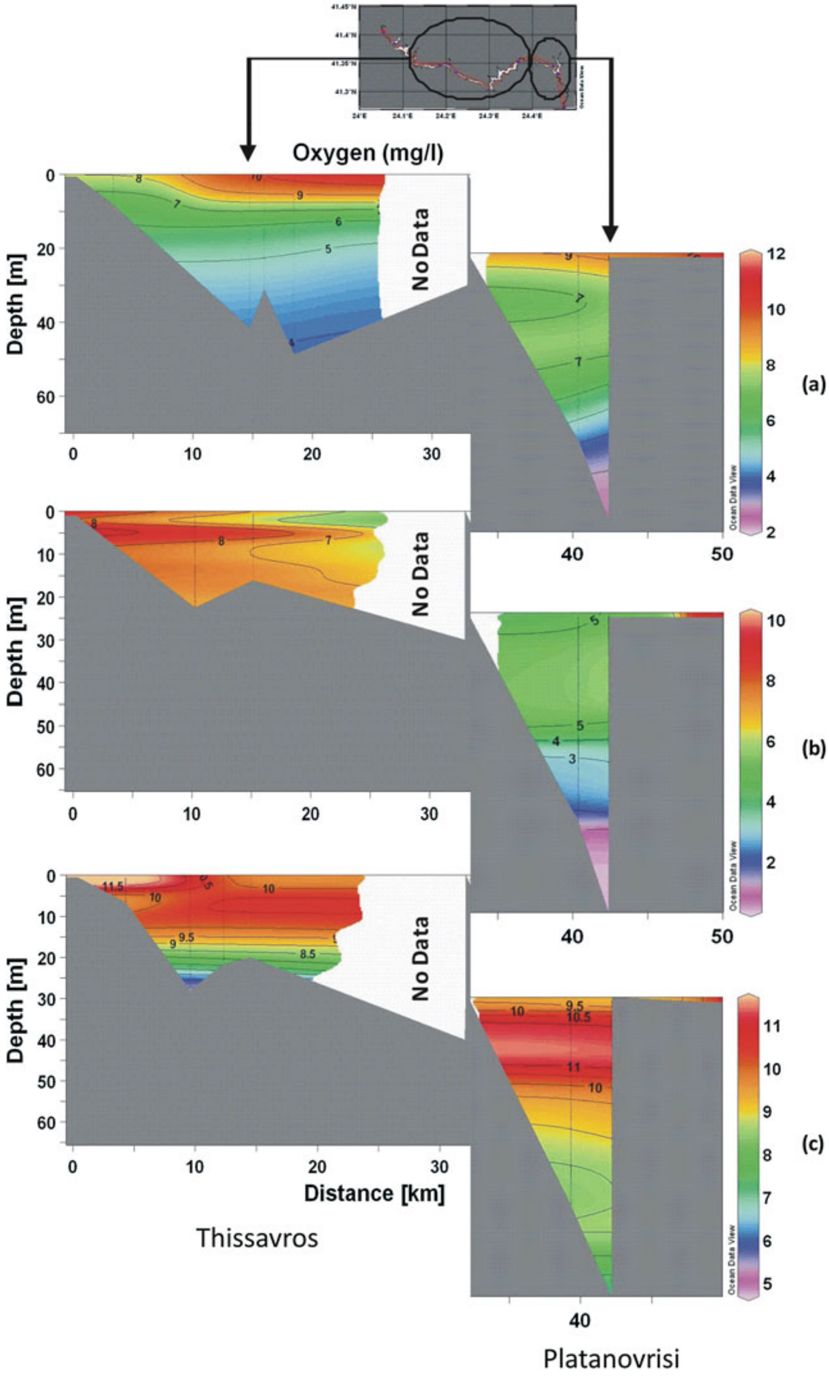


Fig. 3 Dissolved oxygen content (in mg/L) distribution in Thissavros and Platanovrisi Reservoirs during (**a**) summer 2006, (**b**) autumn 2006, and (**c**) spring 2007

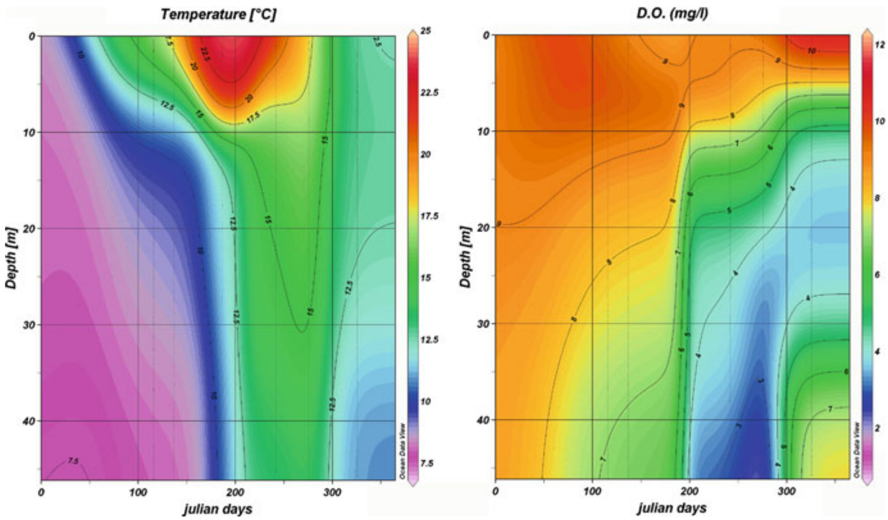


Fig. 4 Temporal variability of (**a**) water temperature and (**b**) dissolved oxygen profiles in Platanovrisi Reservoir during 2007

been regularly reported by local Greek authorities. Such process accumulates dissolved and suspended pollutants (organic matter, nutrients, heavy metals, and toxic substances) into the upstream part of the reservoir, changing water physico-chemical characteristics.

Kamidis [41] monitored the distribution of total suspended solids (TSS) upstream, along both reservoirs and downstream of the dams in Nestos River, revealing that approximately 86–92% of the suspended material is entrapped by the dams (Fig. 5). Summer TSS distributions encountered higher concentrations near Thissavros reservoir bottom, most probably due to the resuspension of sediments under the anoxic hypolimnetic conditions. Hrissanthou [44] estimated a sediment supply reduction in relation to historic sediment yields, at the deltaic zone of 60%, as a result of river damming, while Andreadaki et al. [45] estimate this reduction at 84%.

This entrapment reduces the sediment deposition at the downstream river channel and floodplains, thus affecting river's morphology. This process influences the seashore sediment balance, resulting in an obvious erosion of Nestos River mouth and the adjacent coastline. Xeidakis and Delimani [46] and Xeidakis et al. [47] showed that the Nestos coastal zone is eroded with a rate ranging from a few centimeters up to 25 m/year. Comparing satellite images over the 1982–2001 period, Tsihrintzis et al. [48] estimated a total erosion area of 1.16 km^2 and an accretion area of 0.22 km^2, indicating the net sediment deficit.

Material entrapment and burial mostly occurs along the upper part of Thissavros Reservoir, where sediments appear strongly enriched in cadmium, in contents three times higher than the downstream river, in copper, at approximately six times

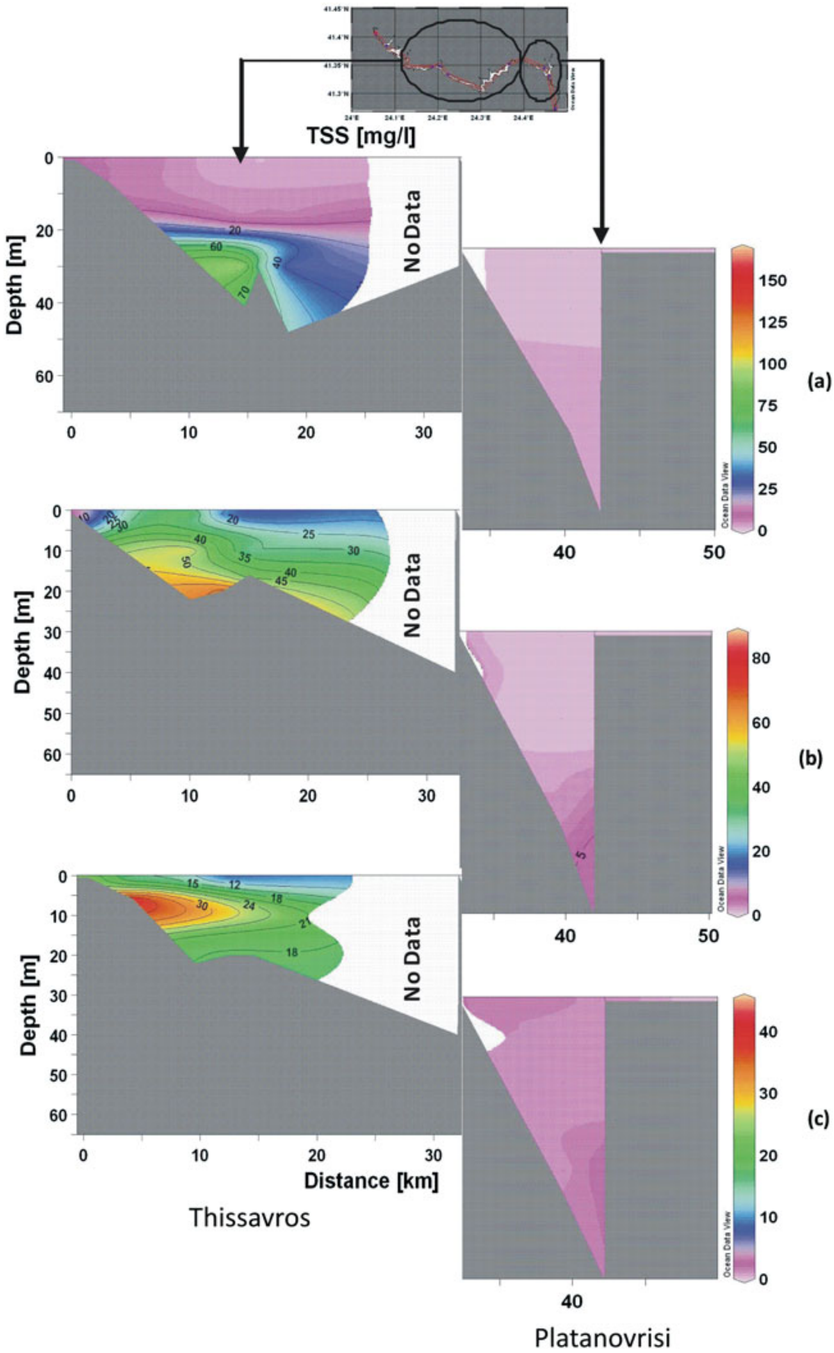


Fig. 5 Total suspended solids (TSS, in mg/L) distribution in Thissavros and Platanovrisi Reservoirs during (**a**) summer 2006, (**b**) autumn 2006, and (**c**) spring 2007

higher, and in chromium and mercury, at approximately ten times higher than the downstream part [41].

Similarly, high nutrient concentrations have been observed at monitoring stations close to the Greek–Bulgarian borders, ranging between 20 and 40 μM for nitrates, 0.6–1.1 μM for nitrites, 4–6 μM for ammonium, and 3–4 μM for phosphates [41]. These nutrients appear concentrated at the bottom of Thissavros Reservoir, as a result of flow reduction and system alteration from riverine into lacustrine. Platanovrisi Reservoir exhibits really low nutrient levels throughout the year, as a result of nutrient filtering at the upstream lake. These changes affect the biogeochemical cycle of systems, altering nutrient relative proportions and affecting phytoplankton production in the system.

Finally, the impact of flow blockage is closely related to the geographic compartmentalization of the river course, obstructing the movement of migratory fishes and genetically isolating aquatic populations. In Nestos River, Thissavros and Platanovrisi Dams have heights of 175 m and 95 m, respectively, making extremely costly any provisions for free fish movement through fish ladder and/or fish passage. Koutrakis et al. [49] identified the problem and studied extensively the impact of Nestos River damming on fish fauna isolation, especially for migratory species as the rainbow trout (*Oncorhynchus mykiss*) and the Nestos trout (*Salmo macedonicus*).

3.3 Impacts Related to Flow Storage

Flow storage in the dam reservoir produces a lacustrine environment changing completely the riverine characteristics of the system. Under such conditions, the impact of the incident solar heat radiation promotes the spring and summer vertical layering and water column thermal stratification, changing significantly the distribution of water physico-chemical properties (temperature, nutrients, DO, etc.). Thermal stratification and destratification events, exhibiting seasonal patterns as governed mostly by local meteorology (solar radiation enhances stratification, while strong winds produce mixing), affect directly the hypolimnetic releases of water from the reservoir towards the downstream part of the river.

Sylaios et al. [42] and Kamidis [41] explained that the presence of a water outlet at the hypolimnetic layer of Platanovrisi Reservoir reduces the water temperature at the downstream part of the river. Such hypolimnetic cold water releases appear more eminent during the summer period, when the upstream-to-downstream water temperature difference reaches its maximum at 11°C (Fig. 6). Cold hypolimnetic releases affect the downstream river habitat, as fish and benthic organisms live under limited seasonal temperature fluctuations. Koutrakis et al. [49] showed a reduction in the downstream faunal richness along the main Nestos River channel, directly related to the cold water releases and the sudden change in water flow conditions.

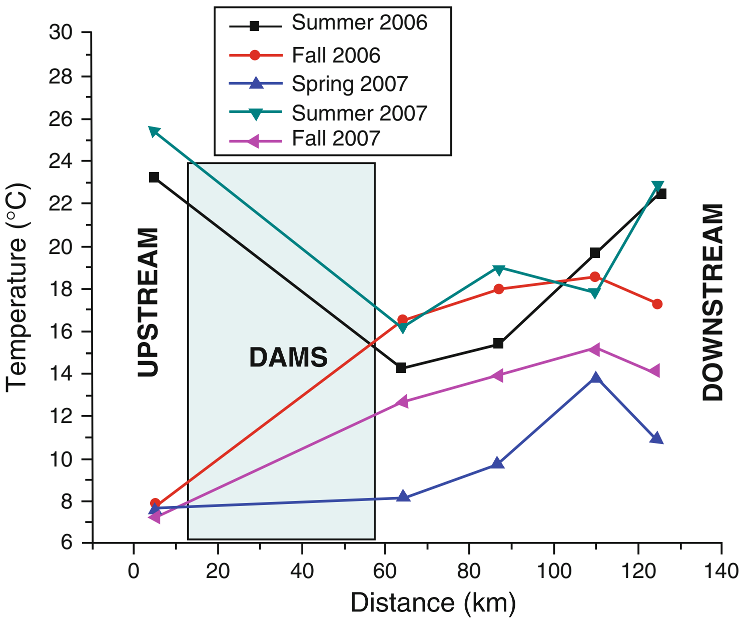


Fig. 6 Change of water temperature upstream and downstream of dam reservoirs in Nestos River

The water re-entering the river at the dams' downstream portion has completely different stoichiometric properties, thus affecting the aquatic biodiversity of the river channel. Kamidis [41] showed that the N:P ratio along Nestos River changes from pure nitrogen limitation (N:P ~ 4–5) at the upstream river part, into phosphorus limitation (N:P ~ 25–30) downstream, as a result of water storage at the reservoirs. This change of water stoichiometry is also associated with a reduction in the nutrients flux delivered to the coastal zone.

3.4 Impacts Related to Flow Regulation

Dam's daily operation leads to the regulation of flow at the downstream river, associated with frequent and rapid water flow changes. Such alterations affect the timing and duration of peak flows (hydropeaking), since these are related to the energy requirements and not to natural causes. Boskidis [50] and Boskidis et al. [51] presented results from six water flow and stage monitoring stations installed along the downstream Nestos River. Water discharge at the dam's nearest station shows a mean value of 37.2 m^3/s, with significant fluctuation up to 220 m^3/s and standard deviation of 44.6 m^3/s (Fig. 7). Daily fluctuations in river stage are also related to dam's operation and the peak energy demand (Fig. 8). Boskidis et al. [52] calibrated and successfully validated a SWAT model in terms of water flows and stage

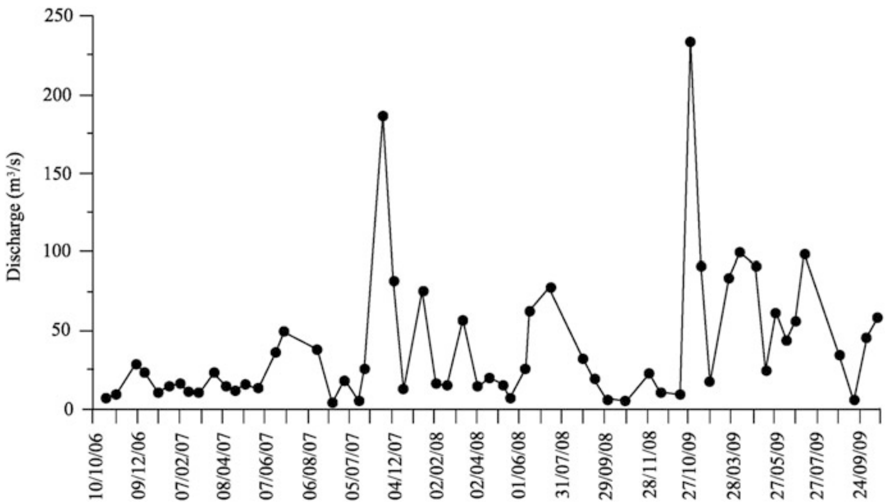


Fig. 7 Temporal variability of Nestos River discharge at the closest station downstream of Platanovrisi Reservoir (Stavroupolis)

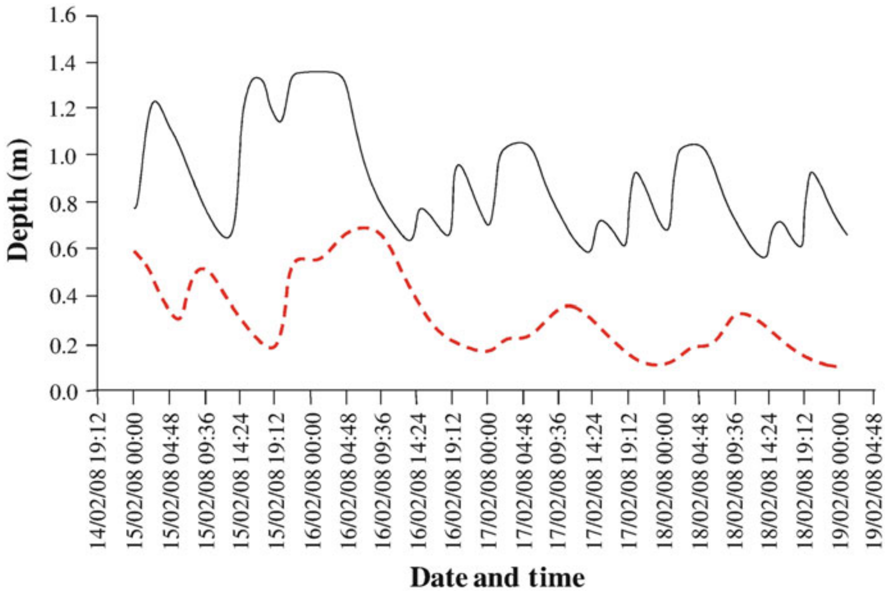


Fig. 8 Temporal variability of water level downstream of Platanovrisi Reservoir (*black line* Stavroupolis; *red line* Galani)

variability, for the downstream Nestos River watershed. The model, with a time-increment of 10 min, simulated these daily flooding peaks showing good agreement with observations. Such flooding fluxes may increase the erosion at the river banks and beds, thus elevating the concentration of TSS [41], and changing channel's morphology downstream.

Flow regulation appears also responsible for variations in the distribution of freshwater fish lengths and ages observed between the upstream and downstream parts of the river [53]. These changes may be attributed to the loss of feeding and reproduction habitats for most species, due to the sudden flushing events occurring daily downstream [54].

Sediment entrapment and water flow regulation directly affect the lowland located Nestos riparian forest, causing functional impairment and serious degradation. Emmanouloudis et al. [55] investigated the sediment dynamics in the bed and delta of the Nestos River, before and after the construction of the dams using satellite imagery and GIS modelling, demonstrating that declining water supply and sediment fluxes led to significant adverse effects to this exceptional system.

Reduction of water discharge at the river mouth leads to a consequent decrease in the area affected by the river plume. Sylaios et al. [56] applied numerical models to examine the impact of river damming on freshwater advection and stratification conditions during the pre- and post-damming periods. Results showed that plume expansion is significantly diminished, affecting the zone limited to the vicinity of the river mouth, while water column stratification, determined by the ϕ_{Total}-term, appears reduced by 50.2% between the two regimes.

Further, such regime change significantly affects the functions of coastal lagoons located at the Nestos deltaic zone. These lagoons are fishery-exploited with a euryhaline fish production of 150 kg/ha/year, reduced strongly lately by 30–40% as a result of freshwater shortage and transfer of agricultural residues. Under such adverse conditions, phytoplankton and macrophytic blooms occur regularly in spring and summer. Bloom occurrence has been successfully predicted for operational purposes utilizing data from on-line monitoring stations and a developed fuzzy-logic numerical model [57].

4 Nestos Watershed Restoration Using an Ecohydrological Approach

Ecohydrology is a rather new concept in ecosystems management (since 1990s), in which the hydrological processes are integrated and interlinked with the ecological, and vice versa, in order to buffer man-made impacts with the ultimate goal of preserving, enhancing, or restoring the capacity of basin's aquatic ecosystems for sustainable use [58]. Through the knowledge of these two processes, researchers attempt to find and apply innovative solutions, at river basin level, aiming to improve the assimilative capacity of the ecosystem by regulating its ecological processes [29]. In that sense, ecohydrology provides a new integrated way of thinking, encompassing the sectoral knowledge of hydrology, ecology, socio-economics, and law into a rigid interdisciplinary context seeking for sustainable environmental solutions [59].

Especially, in heavily exploited transboundary rivers, as Nestos River in Northern Greece, the need for integrative thinking in adopting a dual regulation scheme between hydrology and ecology may mitigate the above-described adverse environmental impacts related with river damming and water abstraction, by enhancing the resilience and adaptation of such system [60]. Such sustainable solutions appear particularly critical in facing the potential challenges of climate change in these vulnerable Mediterranean watersheds.

To mitigate the impact of thermal stratification and the related effects of bottom anoxia and cold hypolimnetic releases from the lower Nestos River reservoir (Platanovrisi), artificial mixing is proposed as the most appropriate ecohydrologic measure. Air bubble diffusers installed near the dam hypolimnetic outlet are expected to induce thermocline destabilization, promoting vertical masses exchange and leading to efficient mixing and oxygenation (Fig. 9). Kamidis et al. [62] applied a 3D hydrodynamic model to study the impact of a series of air bubble diffusers reducing vertical temperature differences at the first 20 m from 15.3 to 3.3°C.

Mixing changes the distribution of temperature vertically, as well as the availability of light and nutrients in the water column. Such method has been widely used to control lake algal blooms and the selective inhibition of cyanobacteria [63]. Becker et al. [61] reported the physical, chemical, and ecological changes induced in the water column of a hypertrophic reservoir, as a result of artificial mixing.

At the downstream part, river flow and stage regulation made most fish species to leave the main river course and inhabit the torrents and tributaries of the system. Electrofishing and artificial reproduction of endangered fish species was performed by the Fisheries Research Institute team [49], within the framework of an Interreg IIIA/PHARE CBC Greece–Bulgaria program. Shallow basins of higher temperature and lower flow conditions were constructed at three sites along river banks to acclimatize and protect the enriched fish populations. Altered hydrologic conditions led to the improvement of ichthyofaunal indices along river's downstream part.

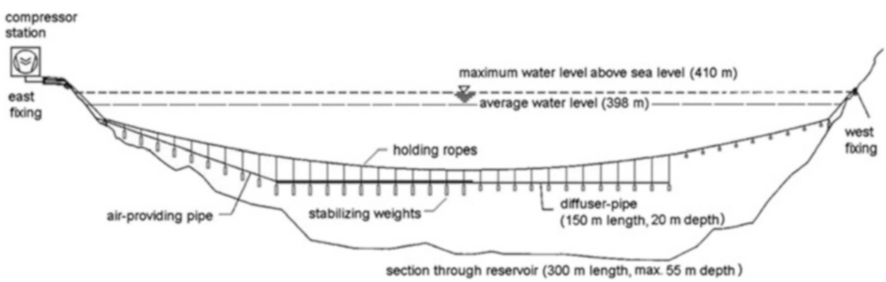


Fig. 9 Artificial mixing system composed by air diffusers and aeration pipes installed near the dam hypolimnetic outlet (after [61])

At the deltaic zone, the old Nestos River branches were reconnected to the main course of the river and re-flooded within the framework of a LIFE-Nature program (Fig. 10). These actions aimed to reduce the seawater intrusions in the mouth area, to create a core protection site without human activities and to provide the opportunity for some rare wildlife species to stabilize and increase their breeding and wintering populations. After reconnection, re-plantation, and reforestation, the water quality in remote surface ponds was improved, while some important priority riparian forest habitats were recovered [64].

At the periphery of Vassova and Eratino lagoons, surface flow constructed wetlands were developed to mitigate eutrophication in their basins by removing nitrogen and phosphorus compounds from agricultural drainage [65]. Indeed, for many decades these lagoons suffered from low dissolved oxygen levels and adverse environmental conditions, resulting in a decline in fish production. The operation of these wetlands achieved to reduce the imported total inorganic nitrogen by almost 85%, total phosphorus by 15%, and total coliforms by 90%, but failed to reduce the inflow of organic matter (BOD and COD).

Further, Nestos Lagoons management to mitigate eutrophication appears directly related to the hydrodynamics prevailing at their tidal inlet. As the geometric characteristics of these inlets are man-controlled, a simple one-dimensional model was developed, adapted, calibrated, and verified to link inlet geometry to lagoon exchange capacity and therefore to its ecology and diversity [66]. The model

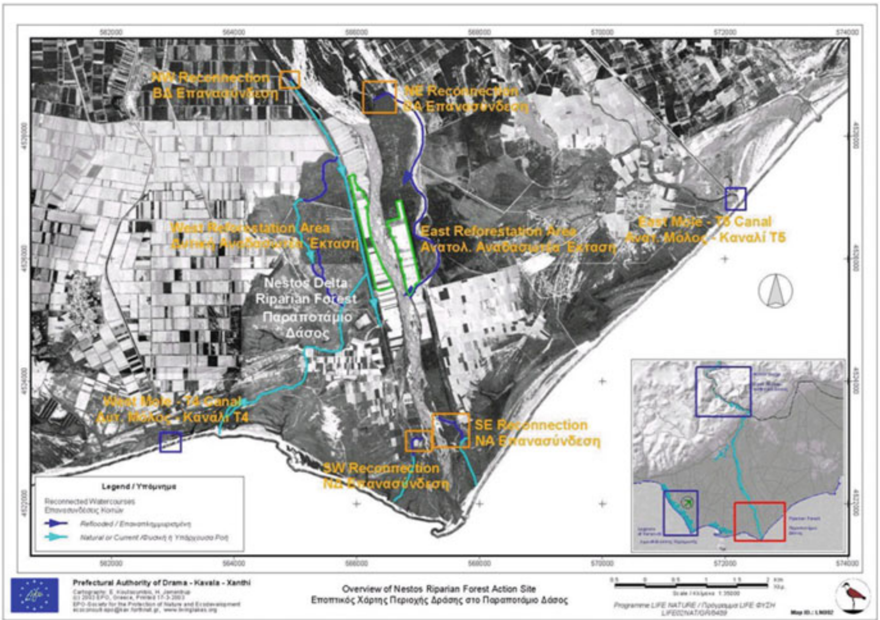


Fig. 10 Reconnection of Nestos River old branches to reduce sea intrusion and improve the deltaic ecosystem functioning

tested several restoration alternatives to improve tidal flushing in parallel to increasing internal and inlet flow at levels that favor the recruitment of juvenile fish.

At the coastal zone, the restoration of the coastline from the induced erosive trend was proposed through an extended beach nourishment project [67]. Submarine relict sand deposits were detected in the vicinity of Nestos River mouth, to be used for potential beach nourishment works. Under various scenarios it occurred that the excavated sediment volume could vary from 5.0×10^6 to 4.3×10^7 m^3. For the restoration of 21 km coastline, these scenarios correspond to a nourished beach width of 52–450 m, respectively, that could sustain the sand (with the present sediment transport rates) for a period of up to 30 years. However, consideration of environmental impacts showed that sea bottom dredging activities and the produced turbidity plumes may affect the benthic assemblages in the sand excavation zone, such as small crustaceans, shrimps, and demersal fishes.

5 Conclusions

An extensive review of the environmental impacts induced on a river watershed by the construction and operation of large hydropower dams is presented herein. Such impacts are expected to affect gradually a higher number of riverine systems, as the needs for freshwater for food and energy usage will increase sharply. Impacts are presented and discussed through the changes seen within a period of 15 years after the construction of two dams along Nestos River, a transboundary watershed located in Northern Greece. Present analysis depicted that these extensive changes are related to reservoirs' filling, river flow blockage, river flow storage, and flow regulation during dams' operation. The newly introduced concept of ecohydrology and the adoption of its management principles could be considered for the mitigation and restoration of these heavily modified systems.

References

1. Nilsson C, Reidy CA, Dynesius M, Revenga C (2005) Fragmentation and flow regulation of the world's large river systems. Science 308:405–408
2. Millennium Ecosystem Assessment (2005) Ecosystems and human well-being: a framework for assessment. Island Press, Washington, DC
3. US Energy Information Administration (2013) International energy outlook. US Department of Energy, Washington, DC
4. Vörösmarty CJ, Sharma KP, Fekete BM, Copeland AH, Holden J, Marble J, Lough JA (1997) The storage and aging of continental runoff in large reservoir systems of the world. Ambio 26:210–219
5. World Commission on Dams (2000) Dams and development: a new framework for decision-making. The report of the World Commission on Dams. World Commission on Dams and Earthscan Publications, London

6. Bird J, Wallace P (2001) Dams and development – an insight to the report of the World Commission on Dams. Irrig Drain 50:53–64
7. European Environmental Agency (2013) Adaptation in Europe: addressing risks and opportunities from climate change in the context of socio-economic developments. EEA, Copenhagen
8. Postel S, Richter B (2003) Rivers for life: managing water for people and nature. Island Press, Washington, DC
9. Brismar A (2004) Attention to impact pathways in EISs of large dam projects. Environ Impact Assess Rev 24(1):59–87
10. Collier M, Webb RH, Schmidt JC (1996) Dams and rivers: a primer on the downstream effects of dams. US Geological Survey Circular 1126, Arizona
11. Lin Q (2011) Influence of dams on river ecosystem and its countermeasures. J Water Res Prot 3:60–66
12. Lerer LB, Scudder T (1999) Health impacts of large dams. Environ Impact Assess Rev 19:113–123
13. Tajziehchi S, Monavari SM, Karbassi AR, Shariat SM, Khorasani N (2013) Quantification of social impacts of large hydropower dams – a case study of Alborz Dam in Mazandaran Province, Northern Iran. Int J Environ Res 7(2):377–382
14. Assani AA, Stichelbout E, Roy A, Petit F (2006) Comparison of impacts of dams on the annual maximum flow characteristics in three regulated hydrologic regimes in Quebec (Canada). Hydrol Process 20:3485–3501
15. Snoussi M, Kitheka J, Shaghude Y, Kane A, Arthurton R, Le Tissier M, Virji H (2007) Downstream and coastal impacts of damming and water abstraction in Africa. Environ Manag 39:587–600
16. Magilligan FJ, Nislow KH (2005) Changes in hydrologic regime by dams. Geomorphology 71:61–78
17. Sanches PV, Nakatani K, Bialetzki A, Baumgartner B, Gomes LC, Luiz A (2006) Flow regulation by dams affecting ichthyoplankton: the case of the Porto Primavera Dam, Parana River, Brazil. River Res Appl 22:555–565
18. McGinnis DF, Bocaniov S, Teodoru C, Friendl G, Lorke A, Wuest A (2006) Silica retention in the iron gate I reservoir on the Danube River: the role of side bays as nutrient sinks. River Res Appl 22:441–456
19. Cravo A, Madureira M, Felıcia H, Rita F, Bebianno MJ (2006) Impact of outflow from the Guadiana River on the distribution of suspended particulate matter and nutrients in the adjacent coastal zone. Estuar Coast Shelf Sci 70:63–75
20. Graf WL (2006) Downstream hydrologic and geomorphic effects of large dams on American rivers. Geomorphology 79:336–360
21. Klaver G, van Os B, Negrel P, Petelet-Giraud E (2007) Influence of hydropower dams on the composition of the suspended and riverbank sediments in the Danube. Environ Pollut 148:718–728
22. Kummu M, Varis O (2007) Sediment-related impacts due to upstream reservoir trapping, the Lower Mekong River. Geomorphology 85:275–293
23. Walling DE (2008) The changing sediment loads of the world rivers. Annals of Warsaw University of Life Sciences–SGGW. Land Reclam 39:3–20
24. Jiao N et al (2007) Ecological anomalies in the East China Sea: impacts of the three Gorges Dam? Water Res 41:1287–1293
25. Anderson EP, Pringle CM, Rojas M (2006) Transforming tropical rivers: an environmental perspective on hydropower development in Costa Rica. Aquat Conserv Mar Freshwat Ecosyst 16:679–693
26. Habit E, Belk MC, Parra O (2006) Response of the riverine fish community to the construction and operation of a diversion hydropower plant in central Chile. Aquat Conserv Mar Freshwat Ecosyst 17:37–49

27. Champalbert G, Pagano M, Sene P, Corbin D (2007) Relationships between meso- and macro-zooplankton communities and hydrology in the Senegal River Estuary. Estuar Coast Shelf Sci 74:381–394
28. Faria A, Morais P, Chıcharo MA (2006) Ichthyoplankton dynamics in the Guadiana estuary and adjacent coastal area, South-East Portugal. Estuar Coast Shelf Sci 70:85–97
29. Zalewski M (2000) Ecohydrology – the scientific background to use ecosystem properties as management tools toward sustainability of water resources. Guest Editorial Ecol Eng 16:1–8
30. Zalewski M (2012) Ecohydrology – process oriented thinking for sustainability of river basins. Ecohydrol Hydrobiol 12(2):89–92
31. Bratrich C, Truffer B, Jorde L, Markard J, Meier W, Peter A, Schneider M, Wehrli B (2004) Green hydropower: a new assessment procedure for river management. River Res Appl 20:865–882
32. Wolf A, Natharius J, Danielson J, Ward B, Pender J (1999) International river basins of the 21 world. Int J Water Res Dev 15(4):387–427
33. Ganoulis J, Skoulikaris H, Monget JM (2008) Involving stakeholders in transboundary water resource management: the Mesta/Nestos 'HELP' basin. In: International symposium 'HELP in action: local solutions to global water problems–lessons from the South', Emperor's Palace, Johannesburg, South Africa, 4–9 Nov 2007
34. Zagorchev I (2007) Late Cenozoic development of the Strouma and Mesta fluviolacustrine systems, SW Bulgaria and northern Greece. Quat Sci Rev 26:2783–2800
35. Diadovski IK, Atanassova MP, Ivanov IS (2007) Integral assessment of climate impact on the transboundary Mesta River flow formation in Bulgaria. Environ Monit Assess 127:383–388
36. Darakas E (2002) The transboundary River Nestos and its water quality assessment: cross-border cooperation between Greece and Bulgaria. Environmentalist 22:367–375
37. Koutroumanidis T, Sylaios G, Zafeiriou E, Tsihrintzis VA (2009) Genetic modeling for the optimal forecasting of hydrologic time-series: application in Nestos River. J Hydrol 368:156–164
38. Kioutsoukis I, Rapsomanikis S, Loupa R (2006) Robust stochastic seasonal precipitation scenarios. Int J Climatol 26:2077–2095
39. Sylaios GK, Bournaski E (2009) The transboundary Nestos/Mesta River: an overview on hydrology and environment. In: Economidis P, Koutrakis M, Apostolou A, Vassilev M, Pehlivanov L (eds) River Nestos fish Fauna. Interreg IIIA, Greece
40. Stoichev K (1989) Fluctuations concerning the annual stream flow in the Rhodope massif. Geogr Rhodopica 2:158–169
41. Kamidis N (2011) Description and simulation of Nestos River plume – investigation of impacts on estuarine ecosystems. PhD thesis, Department of Environmental Engineering, Democritus University of Thrace, Thrace, 393 pp
42. Sylaios G, Tsolkas S, Kamidis N, Boskidis I, Tsihrintzis VA (2009) Temporal variability of water quality in Platanovrisi Reservoir related to stratification/mixing conditions. In: 2nd Conference on Environmental Management, Engineering, Planning and Economics (CEMEPE), vol 1, pp 389–394
43. Koutrakis ET, Sapounidis A, Apostolou A, Vassilev M, Pehlivanov L, Leontarakis P, Tsekov A, Sylaios G, Economidis PS (2013) Impact of dams on freshwater fish fauna composition: the case of a transboundary river. J Biol Res 20:326–338
44. Hrissanthou V (1999) Erosion, transport and sedimentation in the Nestos River Basin. In: Proceedings of international conference on hydrology & geochemistry. Processes of large scale river basins, Manaus, Brazil, 15–19 Nov 1999
45. Andreadaki M, Hrissanthou V, Kotsovinos N (2008) Calculation of sediment reduction at the outlet of Nestos river basin due to the dams. In: IV international symposium on transboundary waters management, Thessaloniki, 15–18 Oct 2008
46. Xeidakis GS, Delimani P (1999) Coastal erosion problems in the Northern Aegean Coastline: the case of Nestos River Delta and the adjacent coastlines. In: Brebbia CA (ed) Coastal environment, environmental problems in coastal regions IV. WIT Press, Ashurst, Southampton pp. 151–159

47. Xeidakis GS, Georgoulas A, Kotsovinos N, Varragouli E (2010) Human interventions and degradation of the coastal zone of Nestos River Delta, North Aegean Sea, N. Greece. In: Williams AL, Pinches GM, Chin CY, McMorran TJ, Massey CI (eds) Geologically active: proceedings of the 11th IAEG congress, Auckland, New Zealand, 5–10 Sept 2010, pp 797–804
48. Tsihrintzis VA, Sylaios G, Akratos C, Pisinaras V (2008) Beach nourishment studies along the East-Macedonia and Thrace coastline. In: Proceedings of the IX international conference on protection and restoration of the environment, Kefalonia, Greece
49. Koutrakis ET, Sylaios G, Kamidis N, Markou D, Sapounidis A, Leontarakis P, Lachouvaris D (2007) Monitoring, protection and strengthening actions for the fish fauna of Nestos River. Final Report of the Interreg IIIA/PHARE CBC program “Actions of management, protection and restoration of fish fauna in Nestos River”, FRI/NAGREF, 97 pp
50. Boskidis I (2010) Environmental management of Nestos River watershed. PhD thesis, Department of Environmental Engineering, Democritus University of Thrace, Thrace, 311 pp
51. Boskidis I, Gikas G, Sylaios G, Tsihrintzis VA (2011) Water quantity and quality assessment of lower Nestos River, Greece. J Environ Sci Health A Tox Hazard Subst Environ Eng 46 (10):1050–1067
52. Boskidis I, Gikas G, Sylaios G, Tsihrintzis VA (2012) Hydrologic and water quality modeling of lower Nestos River Basin. Water Resour Manag 26(10):3023–3051
53. Koutrakis ET (2009) The fish fauna of Nestos River, after the construction of two hydro-power dams. NAGREF Bull:9–14. (in Greek)
54. Sapounidis A, Koutrakis ET, Leonardos ID (2011) Length–weight relationships of 13 species from a flow-regulated Balkan river. J Appl Ichthyol 27:1406–1407
55. Emmanouloudis D, Myronidis D, Panilas S, Efthimiou G (2006) The role of sediments in the dynamics and preservation of the aquatic forest in the Nestos Delta (Northern Greece). ROYAUME-UNI, vol 306. International Association of Hydrological Sciences, Wallingford, pp 214–222
56. Sylaios G, Kamidis N, Tsihrintzis VA (2010) Impact of river damming on coastal stratification–mixing processes: the cases of Strymon and Nestos Rivers, N. Greece. Desalination 250:302–312
57. Sylaios GK, Gitsakis N, Koutroumanidis T, Tsihrintzis VA (2008) CHLfuzzy: a spreadsheet tool for the fuzzy modeling of chlorophyll concentrations in coastal lagoons. Hydrobiologia 610:99–112
58. Zalewski M, Janauer GA, Jolankai G (1997) Ecohydrology. a new paradigm for the sustainable use of aquatic resources. Technical documents in hydrology No. 7. UNESCO, Paris
59. Chapin III FS, Hoel M, Carpenter SR, Lubchenco J, Walker B, Callaghan TV, Folke C, Levin SA, Maler K-G, Nilsson C, Barrett S, Berkes F, Crepin A-S, Danell K, Rosswall T, Starrett D, Xepapadeas A, Zimov SA (2006) Building resilience and adaptation to manage Arctic change. Ambio 35:198–202
60. Sylaios G (2011) Environmental impacts of large-scale hydropower projects and applied ecohydrological solutions for watershed restoration: the case of Nestos River. In: Second conference on healthy rivers and sustainable water resources management, Chongqing, China, 20–22 Nov 2011
61. Becker A, Herschel A, Wilhelm C (2006) Biological effects of incomplete destratification of hypertrophic freshwater reservoir. Hydrobiologia 559:85–100
62. Kamidis N, Koutrakis M, Sylaios G, Triantafillidis S, Tsihrintzis VA (2016) Simulation of the temperature distribution in Thissavros reservoir (River Nestos): investigating the significance of wind action and draft tube mixing on the fate of thermal stratification. In: Proceedings of 16th Panhellenic Conference in Ichthyology, Kavala, Greece
63. Simmons J (1998) Algal control and destratification at Hanningfield reservoir. Water Sci Technol 37:309–316
64. Pergantis F, Dimopoulos P (2006) Reforestation actions at the Nestos River Delta and monitoring within the framework of LIFE 02 NAT/GR/5489. In: LIFE-nature projects and their impact in Greece, pp 45–47

65. Akratos CS, Tsihrintzis VA, Pechlivanidis I, Sylaios GK, Jerrentrup H (2007) A free-water surface constructed wetland for the treatment of agricultural drainage entering Vassova Lagoon, Kavala, Greece. Fresenius Environ Bull 15(12B):1553–1562
66. Tsihrintzis VA, Sylaios GK, Sidiropoulou M, Koutrakis ET (2007) Hydrodynamic modeling and management alternatives in a Mediterranean, fishery exploited, coastal lagoon. Aquac Eng 36:310–324
67. Sylaios G, Anastasiou S, Tsihrintzis VA (2012) Restoration of a seashore eroded due to dam operation through beach nourishment. Ecohydrol Hydrobiol 12(2):123–135

River and Wetland Restoration in Greece: Lessons from Biodiversity Conservation Initiatives

Stamatis Zogaris, Nikolaos Skoulikidis, and Elias Dimitriou

Abstract Rivers in Greece have seen extensive human-induced degradation, and there are increasing demands on the goods and services they provide along with increasing threats from future anthropogenic pressures. These multi-scale alterations to rivers and associated wetlands and riparian zones have severely impacted biodiversity. The Greek government has responded by creating various new protected areas and promoting interest in conservation, while attention to monitoring waters has increased with the implementation of the EU WFD. Unfortunately, bioassessment-based monitoring, long-term conservation programmes and restoration actions in rivers have lagged behind other EU countries. Here we outline the state of river and wetland restoration progress; we describe key restoration examples and discuss shortcomings, pitfalls and opportunities in various aspects of restoration.

Keywords Biodiversity conservation, Environmental history, Restoration, Rivers, Wetlands

Contents

S. Zogaris (✉), N. Skoulikidis, and E. Dimitriou
Institute of Marine Biological Resources and Inland Waters, Hellenic Centre for Marine Research, 46.7 km Athens-Sounio Av., Anavissos 19013, Greece
e-mail: zogaris@hcmr.gr

N. Skoulikidis et al. (eds.), *The Rivers of Greece: Evolution, Current Status and Perspectives*, Hdb Env Chem (2018) 59: 403–432, DOI 10.1007/698_2017_471,

1 Introduction

Restoring rivers is somewhat like a complex medical therapy. Rivers are the "arteries and capillaries of the earth," providing pathways of water, sediments, nutrients and biota, and they are vital to human societies [1]. Rivers nourish and interact with wetlands and riparian zones creating living networks of corridors in the landscape [2]. The restoration of these ecosystems is not a straightforward undertaking; it requires the harmonised engagement of science, policy, local communities and management practices. Restoration does not solely mean returning ecosystems to a natural condition; rather it is an umbrella term to cover the full array of rehabilitation, mitigation, habitat enhancement, species transplants and other restorative measures meant to reduce the impact of human-induced degradation [3–5]. Ecological restoration in rivers should be understood as something concerning broader spaces than aquatic river channels; thus, an integrated river basin management approach should be pursued [6, 7]. A more science-based adaptive management framework takes careful planning and may have high costs. The challenge of scientifically guiding cost-effective restoration may help us to better understand the structure and functioning of complex river-wetland-riparian ecosystems and the management schemes required within specific sociocultural contexts [8].

In Greece, rivers and wetlands have been "managed" since prehistoric times [9, 10]. Some of the oldest reclamation projects, converting wetlands to agricultural land, were in karstic basins such as the lakes of Kopais (Boeotia) and Dystos (Euboea) [11]. Although extensive irrigation projects expanded in the 1950s, there is evidence that even in low-intensity traditional cultural landscapes, waters were often overexploited, long before the modern industrial era [12]. During the last five decades, water use for lowland agriculture has greatly increased, and large-scale water diversions and water storage structures have been created [13, 14]. Yet Greece remains a global hotspot for river and wetland conservation due to its rich aquatic biodiversity. The territory of Greece includes eight distinct freshwater ecoregions; it is a biological crossroad of unique international interest [15]. Remarkably, much of this diversity of river types and conditions is poorly documented, and there are no completed maps that accurately depict natural flow river regimes or a full wetland inventory for the entire country [16, 17]. In this context of data scarcity, long-term human-induced water stress, increasing anthropogenic pressures and threatened biodiversity, conservation and restoration actions may best be considered as a "crisis discipline" [18].

Various reasons have made restoration projects in Greek rivers and wetlands to lag behind in comparison to other EU Mediterranean countries. However, restoration is often mentioned as an important part of biodiversity conservation both in designated protected areas and in river water management units (i.e. the WFD 2000/60 water bodies); yet the restoration proposals are usually left "on paper." In this chapter, we provide a review based on accumulated experience of biodiversity restoration initiatives in rivers and associated wetlands in Greece. We also explore the shortcomings, pitfalls and opportunities in various aspects of restoration and propose some unmet needs and initiatives.

2 A History of River and Wetland Alteration in Greece

More than half of Greece's major river water bodies are now degraded by human-induced pressures [19, 20]. Furthermore, it is estimated that 68% of Greece's wetlands were completely drained or destroyed during the twentieth century [21, 22], while the larger ones endured important human pressures [14, 23]. Despite widespread impoverishment and loss, the concept of river and wetland restoration in Greece is something quite recent and poorly developed. In order to promote actions towards "therapy" for these degraded ecosystems, it is important to understand the historical context that has led to their current state.

There are historical and cultural reasons that explain a significant lag behind other European countries in the conservation and restoration of Greece's rivers and wetlands. Until 1974, malaria was endemic and widespread; Greece is said to have had the highest prevalence of malaria in Europe during the early twentieth century [24]. Wetlands and riparian zones of any kind, including lowland rivers and riparian floodplains, were considered "unhealthy" areas requiring drainage projects and mosquito poisoning campaigns [25, 26]. After WWII, hundreds of small wetlands and river floodplain ecosystems were destroyed or reclaimed to rid the country of these so-called unhealthy conditions. In the Greek islands, many wetlands were in-filled, drained or converted to industrial sites (e.g. many of the Greek island airports are on former wetland sites) [16].

Extensive efforts for agricultural development and land reclamation began in the 1920s and 1930s, in an effort to assist the resettlement of more than one million refugees who flooded in from Asia Minor after the 1919–1922 Greco-Turkish war. After the WWII occupation in the 1940s and the Greek civil war (1944–1949), an intensification of so-called agricultural reclamation and river engineering took place during the 1950s and 1960s [25]. The Greek Military Dictatorship (1967–1974) continued some unusual wetland drainage projects (e.g. the destruction of several wetlands on Euboea, Skopelos, etc.) [16]. Even as late as the early 1980s, wetland drainage and anachronistic river engineering and embankment-building anti-flooding works were ongoing due to efforts for regional economic development (e.g. the draining of Kandila Lake in the Peloponnese in 1981 and the raising of the Kerkini weir on the Strymon in the early 1980s). In one spectacular case, an entire deltaic lagoon was drained by local farmers in the Evros Delta, in 1987 [27]. In the name of supposed flood protection schemes, various river engineering projects, check dams and riparian modification works have been practised even in many upland areas [28] and especially in areas after wildfire damage.

By the late 1990s, many of the larger major wetland and lake areas were officially designated protected areas (i.e. including ten Ramsar sites, many sites in EU Natura 2000 network). As a result, many smaller, isolated wetland sites and small stream environments have been degraded to a greater extent than larger wetlands during the last 25 years [16]. In the rivers, although the rate of large dam building declined, many smaller dams and hydroelectric plants have been developed and are being planned on many small tributaries. The tapping of springs, over-abstraction and water transfer works continue to silently destroy aquatic life in small rivers and streams. Usually the small stretches of perennial waters that existed downstream of springs are given little consideration and are lost after a single water abstraction project. Despite attempts to protect the waters "on paper," as promoted by the EU Nature Directives, the war on "natural waters" has continued into the twenty-first century. Many incremental changes continue to degrade the state of rivers, small wetlands and riparian areas. In this century, some projects, which were supposedly acting as irrigation developments, have also masked actual drainage schemes, even in protected areas; one such case is the Acheron Delta drainage project, where a tunnel was dug through a hill to drain a wetland for agricultural and touristic development in the 2010s [29]. The artificial desiccation during the long summer period and total control of floodwaters aids the agricultural expansion of water-hungry crops in the smaller river deltas such as the Evrotas Delta [30]. These examples reinforce the fact that many smaller sites of lotic and wetland habitat are continually being degraded and much of the loss is poorly documented.

Although irrigation is responsible for approximately 85% of water consumption, adequate restrictions in this sector have not yet been developed and the domestic and industrial sectors are not water efficient [5, 31, 32]. In Mediterranean-type rivers, seasonal pollution is closely related to summer-autumn drought and associated declines in flows and water levels. Nitrogen fertilisers, nitrates, pesticides, phosphorous and organic discharges from urban and agricultural wastewater are the main pollutants of rivers in Greece [33] and these pressures intensify during drought periods. Moreover, mismanagement of irrigation schemes may lead to excessive irrigation water returning to rivers and wetlands which results in an increase of nutrient, sediment and pollution inputs into these systems [34]. Pollution crimes, such as illegal dumping of untreated sewage and industrial wastes directly into rivers and wetlands, are often documented, but polluters are rarely apprehended. The local pollution of groundwater is also a serious problem (e.g. Asopos river). Although the chemical quality of surface waters is generally satisfactory in the uplands, many lowland rivers, even small streams and wetlands, show the effects of seasonal pollution, while their impacts have been poorly monitored up until recently [35]. Implementation of EU legislation and structural funds in the last 25 years have been decisive in developing municipal waste water treatment infrastructure in all major cities, towns and industrial areas [36]. Some sewage treatment plants may periodically malfunction or are poorly planned and/or maintained, but overall, the large number of sewage treatment facilities has helped to ameliorate conditions in some river stretches, especially near certain cities [33].

The effects of drought and other meteorological extreme events, perhaps related to human-induced climate change, have caused significant damage to river and wetland biodiversity. The intensity of near-total desiccation during unusually prolonged summer droughts increases, since during these events, over-abstraction by humans also peaks. During prolonged drought, pollution or salinisation may often intensify [5, 37] and is usually signalled by mass fish deaths (e.g. [38, 39]; and references herein). In many cases, a lot of aquatic species eventually recolonise after the drought [40], but in some cases, certain invertebrates such as freshwater mussels and other molluscs and fish species will not be able to recolonise due to anthropogenic barriers to dispersal (such as dams and weirs). As a result, extirpation of many species, both locally threatened populations and widespread species, is commonplace, although inadequately documented in the literature [41]. One of the most dramatic periods of widespread desiccation took place between 1987 and 1992 in southern Greece [13, 42]. Although organised biological monitoring did not exist during this period, some species of fishes have been lost from certain river basins during this period [42, 43]. The effects of climate change and prolonged drought and ecosystem degradation due to ensuing climate variability and other associated effects on waters are predicted to have severe effects on biodiversity, such as cold-water fish species, in the near future [44].

Apart from the rivers and wetlands, riparian zones have also suffered, but these have been poorly inventoried or delineated in Greece [45]. The effects of riparian degradation are also seen in upland rivers [28] away from the lowland areas where agricultural intensification often erases any trace of riparian vegetation. Much of the damage in riparian areas and riverbank conditions is difficult to document and very widespread [46]. An increase in the roads next to streams and rivers, even within protected areas, has created remarkable damage particularly after the mid-1980s. The degradation of riparian zones in Mediterranean rivers often has powerful and unexpected effects on aquatic ecosystems (water temperature changes, increased siltation, erosion, instream habitat alteration, etc.).

Finally, a kind of "biological pollution" in the form of invasive animal and plant species is also a serious stress and mounting threat for Greece's river basins. Often influenced by changing hydrological conditions, water transfer projects, increased eutrophication and increased artificial storage of waters in reservoirs, many alien species in inland waters are on the rise [47]. Until the early 1990s, Greece had a relatively low incidence of alien fish species, but this number has increased especially in lakes and reservoirs and the lowland areas of the larger river basins [43]. Over 30 alien fish species are now present, some of which are the most invasive and harmful species in Europe, including eastern mosquitofish (*Gambusia holbrooki*), pumpkinseed (*Lepomis gibbosus*), Prussian carp (*Carassius gibelio*) and topmouth gudgeon (*Pseudorasbora parva*). Some species are spread by anglers, in order to increase recreation values; this includes species native to the country but artificially introduced outside the boundaries of their natural distribution. Trout and other native species, which are translocated from one river basin across ecoregional boundaries to other river basins, are a serious form of biological pollution. Although similar looking to the host basin species, these translocated alien fishes may result in hybridisation and genetic introgression (or effects such as outbreeding depression), which could affect reproductive success and cripple local native populations [48]. Furthermore, fish also transport fish diseases and alien invertebrates that can also infect areas where they did not formally exist. These anthropogenic changes could severely alter and/or homogenise previously endemic-rich ecosystems.

3 Conservation Policy and Cultural Context

Between the early 1990s and up until the first decade of this century, Greece increased its designated protected area cover from less than 3% to roughly 27% of the land territory [23, 49]. This was primarily a result of the EU Birds and Habitats Directives, which promoted the Natura 2000 network of protected areas and was supported by many NGOs and members of the scientific community [49]. Along with the WFD, which gives special reference to protected areas, grounds for management and restoration have ameliorated. Since the late 1990s, Greece has witnessed some positive institutional changes that assisted conservation management [50]. However, as is the case in other parts of southern Europe, Greece has struggled with severe implementation problems in its protected area network [51] and management plans [49, 52], and this has stalled progress in effective management, conservation measures and restoration, especially after the Athens 2004 Olympics.

Public awareness for water issues has risen in Greece. It is remarkable that a paradigm shift in many local communities' outlook on the values of river corridors and wetlands has taken place particularly since the early 1990s [5, 26, 53]. Wetlands and river valleys and deltas, once considered unhealthy wastelands, are being promoted for protection and "eco-development" [26, 54]. Although the value of Greece's river deltas and wetlands was originally promoted by visiting naturalists in the 1960s [55], the government commitment essentially began with the first delineation of large wetland areas in the 1980s enacted since Greece's ratification of the Ramsar Convention in 1975 and the Birds Directive's Special Protected Areas designations after 1980 [49]. Afterwards, many Greek NGOs and certain researchers and policymakers were effective in influencing society and state policy with effective protected area creation initiatives [54]. Especially the efforts of environmental NGOs are well known in the remarkable and rapid expansion of preliminary conservation areas or "paper parks" [56]. Much work by environmental NGOs promoted effective wetland protection and on-the-ground initiatives in particular. Few of these campaigns focused on restoration, and the focus was protection; for example, one of the longest and most bitter struggles concerned the Acheloos river transfer ([57] and references herein). Certain NGOs provided outstanding support for long-term conservation actions in certain showcase-protected areas such as Prespa, Zagori and Dadia, often touching on water-related conservation issues. Also, a small number of scientists in government institutions and academia effectively promoted a new interpretation of wetland values and the demand for protection of the larger wetlands and river delta areas [26, 58, 59]. Although problems did exist, conservation and restoration were often politically tied to efforts for EU-funded structural projects, tourism development and local identity promotion (see, e.g. [60–62]).

Efforts for protected area conservation, thus, gained an unprecedented rise in Greece especially after the early 1990s. By the early 2000s, a preliminary system of protected area management bodies was set up for 28 management bodies of the Natura 2000 network, many of them dominated by wetlands and lentic water bodies. Conservation and ecosystem restoration have become targets in the National Biodiversity Strategy [63]. Significant funding for biodiversity conservation developed after the Habitats Directive came into force in 1992. The EU financial instrument for the environment, the LIFE funding mechanism, achieved some important results, and it dominates Greece's aquatic-wetland-riparian restoration history.

4 Review of River and Wetland Restoration

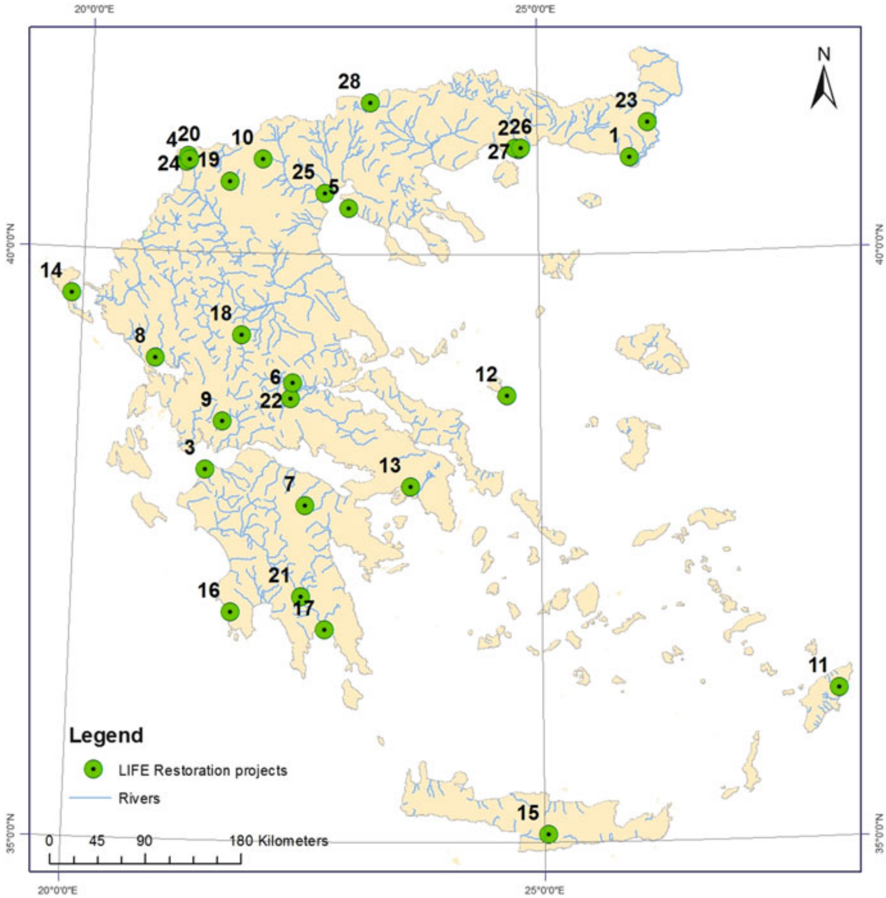
Restoration proposals, actions and monitoring are poorly reported in the scientific literature in Greece [5, 64]. During the 1970s and 1980s, there were no restoration actions aiming at conservation of the natural environment in rivers or wetlands in Greece. Very rarely were restoration works an advocacy issue in early conservation campaigns, which, until the early 1990s, concentrated on antipollution and preservation of natural areas or urban issues [49, 65]. Even urban park and stream engineering works are, by European standards, wholly anthropocentric, and there are few examples of creating or recreating new river habitat for river biodiversity, as was commonplace in many major cities in Europe and North America [66]. Restoration works are notably scarce in small rivers or intermittent and urban streams outside protected areas [45]. Until the mid-1990s, many studies and statements drafting interest for river restoration were made; however, nearly all of these ideas were left on paper.

Major ecological and restoration targeting biodiversity in inland waters effectively began with EU policy-relevant actions under the LIFE initiative in the mid-1990s. Since the inception of the LIFE programme, 213 projects have been implemented in Greece (up until mid-2014), which involve environmental innovation, protection of nature or biodiversity as well as information and communication. These projects correspond to overall investments amounting to €280 million, of which the EE contributed €150 million [63]. A review of the LIFE-funded projects where funds were placed to actively restore or enhance attributes of the river, lake, riparian or wetland environment in Greece within the LIFE initiatives shows that very few projects affect river or lotic ecosystems in general. Twenty-four (24) LIFE project examples are shown in Fig. 1; all of these have some interaction with river sections and associated wetlands. These projects were often of fairly large scale, involving several separate conservation and restoration actions, most targeting species and habitat types that are of priority for conservation in the European Union. Focus on habitat enhancement work for particular bird species dominates the actions. In fact, by 2005 more than 1/3 of all funds given for LIFE-Nature projects in Greece targeted just four high-profile vertebrate species, namely, Mediterranean monk seal, loggerhead turtle, brown bear and bearded vulture [67].

The most effective restoration actions for inland water biodiversity conservation are associated with long-term investments (i.e. 3-year projects or more) in the larger protected areas, such as Prespa, Amvrakikos, Koronia-Volvi, Strofylia-Kotychi, Axios, Nestos Delta and Evros Delta [62, 64]. One such project in the Amvrakikos (1999–2003) created a sluice-controlled break in the artificial embankment of the Louros river to re-wet the former river delta swamp of Rodia [68, 69]. The work was associated with the reintroduction of water buffalo for reed control [70]. The Amvrakikos LIFE project (and an INTERREG project immediately after this) also promoted a restoration of riparian woods along the Louros [71] (Fig. 2). Similar works on various wetland types have taken place in the Evros and Nestos Deltas [64]. The largest riparian forest restoration work in Greece has taken place at the Nestos Delta, where about 80,000 trees have been planted in an area of 280 ha [72, 73]; this initiative is also one of the few well-monitored restoration efforts in the country [74, 75]. Many LIFE projects also promoted lesser actions that affected wetland habitats such as pond building, riparian tree planting and water control structures for management.

Other than the LIFE mechanism, other projects also promoted biodiversity restoration, but these were often of small spatial and temporal scale and are very poorly documented. The EU INTERREG programme and the EU Structural Funds have also assisted restoration efforts particularly in the late 1990s and early 2000s. NGOs, government research institutes and university departments have been active too, often in cooperation with the local and federal government [76]. Some of these small projects were promoted by NGOs and local government, most of them in the 1990s and 2000s. Examples include restoration works such as removing in-filled debris (garbage and construction site infill) from coastal wetlands in Attika [77, 78] and a remarkable project to remove in-filled debris at the Moronis river and river mouth wetland, Souda, Crete [79]. Mitigation measures that promote wetland biodiversity restoration have rarely been practised, apart for the exceptional example in Schinias Marathon National Park in Attika after the Olympic Games [80].

Although conditions were more mature after the year 2000, it is remarkable that few ecological restoration projects were initiated in this century. The 2004 Olympic



No	TITLE	REFERENCE
1	Restoration and conservation management of Drana lagoon in Evros Delta	LIFE00 NAT/GR/007198
2	Habitat Management and Raptor Conservation in Nestos Delta and Gorge	LIFE02 NAT/GR/008489
3	Conservation management in Strofylia-Kotychi	LIFE02 NAT/GR/008491
4	Conservation of priority bird species in Lake Mikri Prespa, Greece	LIFE02 NAT/GR/008494
5	Actions for the conservation of coastal habitats and significant avifauna species in NATURA 2000 network sites of Epanomi	LIFE09 NAT/GR/000343
6	Conservation of priority forests and forest openings in "Ethnikos Drymos Oitis" and "Oros Kallidromo" of Sterea Ellada	LIFE11 NAT/GR/001014
7	Sustainable management and financing of wetland biodiversity – The case of Lake Stymfalia	LIFE12 NAT/GR/000275
8	Conservation management of Amvrakikos wetlands	LIFE99 NAT/GR/006475
9	Actions for the protection of the calcareous fens	LIFE99 NAT/GR/006499
10	Implementation of management measures at the Agras wetland	LIFE03 NAT/GR/000092
11	Conservation measures for the endangered fish Ladigesocypris ghigii	LIFE98 NAT/GR/005279
12	Demonstration of the Biodiversity Action Planning approach, to benefit local biodiversity on an Aegean island, Skyros	LIFE09 NAT_GR_000323
13	Integrated approach for solving the problem of liquid hydrocarbons present in the Hellenic Aspropyrgos refinery (HAR)	LIFE96ENV/GR/000535
14	A Resource Exchange Programme For River Potamos	LIFE98 ENV/GR/000234
15	Mediterranean reservoirs and wetlands. A demonstration of multiple - objective management in the island of Crete	LIFE00 ENV/GR/000685
16,17	Implementation of management plan for Pylos Lagoon and Evrotas Delta	LIFE97 NAT/GR/004247
18	Implementation of management actions for Tavropos Lake area in Greece	LIFE99 NAT/GR/006480
19	Conservation management of Cheimaditida-Zazari wetlands	LIFE00 NAT/GR/007242
20	Bird conservation in Lesser Prespa Lake: benefiting local communities and building a climate change resilient ecosystem	LIFE15 NAT/GR/000936
21	Development of Ecotourism in the Riparian Ecosystem of Evrotas near Sparta	LIFE94 ENV/GR/001263
22	Integrated management of Sperchios River ecosystem	LIFE92 ENV/GR/000054
23	Conservation of birds of prey in the Dadia Forest Reserve, Greece	LIFE02 NAT/GR/008497
24,25,26	Conservation of Phalacrocorax pygmaeus and Anser erythropus in Greece	LIFE96 NAT/GR/003217
27,28	Living Lakes: Sustainable Management of wetlands and shallow lakes	LIFE00 ENV/D/000351

Fig. 1 Examples of conservation-oriented EU LIFE-funded projects that involved restoration practices in Greece's rivers and associated wetlands in the last 20 years

Fig. 2 Restoration works at Amvrakikos wetlands (clockwise from *top left*): the river Louros floodplain at Petra (riparian forest restoration actions); canal and sluice construction in the Louros floodplain (2001); reintroduced water buffalo to enhance reed-bed diversity; canal reconnecting to wetlands and buffalo fencing (Photos: S. Zogaris)

Games was an important catalyst for various mega-construction projects in Athens, and some ideas and mature proposals that were meant for conservation were actually displaced by pressing Olympic Games infrastructure timelines after late 1997, the year that the Games were awarded to Athens. After 2008, work on restoration also subsided due to Greece's economic recession. The economic crisis years helped shift the attention away from nature conservation and restoration aims which had been proposed since the 1990s [50]. A prime example of this is the notable delay in the application of the programme of measures in designated river water bodies for the implementation of the EU Water Framework Directive 2000/60/EC in Greece.

5 Achievements

5.1 Inventory and Classification Baselines

In-depth wetland and water body inventories were first introduced in the early 1990s [81], and they continue with policy-relevant applications. Baseline wetland site inventory work was done by the Greek Biotope-Wetland Centre (EKBY) [82] and by WWF Greece [16]. River, lake and riparian zone monitoring was promoted by HCMR and EKBY [17, 20]. Typologies for artificial water bodies and their values to biodiversity were also recently attempted [83]. These baselines are of outstanding value for conservation planning and are all fairly recent developments that require refining and completion [17]. These efforts were also important in providing the first surveys of Greece's richness in terms of all inland water ecosystems. They also assisted in promoting policy stakeholder and public sensitisation.

5.2 Local Studies and Restoration Planning

Due to EU policy directives and land use planning needs, many biodiversity conservation studies were commissioned particularly between the early 1990s and the first decade of the 2000s. Restoration was often mentioned and designs were set [84]. Some challenging issues such as artificial water bodies have also recently been investigated (e.g. reservoirs on Crete, see [85]). Proposed management plans within the policy-relevant special environmental studies for delineating and applying measures for protected areas provide many practical experiences [64]. In many cases, large-scale studies focused primarily on severe degradation problems in lakes and coastal lagoons (e.g. Lakes Pamvotis, Kastoria, Koumoundourou, Karla, Koronia) [86–88]. River basin management in an integrated form is being developed, and science-based management is being promoted in recent years, often incorporating needs for restoration (e.g. [84, 89]). Many of the studies have become interdisciplinary in recent years, but research and proposals continue to focus on lentic rather than lotic water bodies (e.g. [90]).

5.3 Learning by Practice

Biodiversity conservation-oriented restoration actions, particularly through the EU LIFE projects, focused on threatened species and, in particular, the so-called priority species and habitat types, as demanded by relevant EU directives. These included emblematic ecosystems such as lakes, coastal lagoons, riparian woods and various river delta habitats and wetlands. Efforts targeting habitats in places such as the Nestos Delta, Evros Delta, Amvrakikos Gulf wetlands, Gialova-Pylos, Strofylia-Kotychi wetlands and Prespa are in tune with local conservation awareness and should be regarded as important "initiating acts" for restoration [64, 91]. Much of this often complicated, restoration-conservation work had never before been attempted in Greece, so it is an important "practice" for future work. In fact, it has been said that without the LIFE-funding mechanism, modern biodiversity conservation practice, as we know it, would be "non-existent" in Greece! [92].

However, there are many biodiversity elements that have still received very little attention. Fish are rarely targeted by conservation-restoration projects, and there is very little experience in restoring the ichthyofauna in Greece. In the Spercheios river basin, one small government-funded conservation project successfully executed the first species transfer (assisted migration) of the endemic Greek stickleback *Pungitius hellenicus* to an adjacent spring-fed pool that harboured no fish in 1997 [93]. The small population of translocated fish survives to this day. Efforts for translocation of similarly range-restricted species have also recently begun in other parts of Greece (Kalogianni, E. pers. com). Another important fish-based project, this time a LIFE-Nature project, targeted the endemic Gizani of Rhodes, *Ladigesocypris ghighii*, and was also important as an initiation into river restoration actions for a threatened freshwater fish in Greece [94]. During the project's survey work, the threatened species was also discovered in new areas on the island [95]. Remarkably, no other LIFE project has applied integrated, scientifically led, restoration targeting endangered freshwater fish species in Greece [96].

Since Mediterranean riverine landscapes and riparian zones have seen a decrease in ungulate grazing regimes, the reintroduction of grazing by large herbivores is important. This kind of work was initiated in wetlands in the late 1990s. In Greece, reintroduction of water buffalo has seen success [97, 98]. Good evidence-based restoration results were produced during a LIFE project also in reed-bed management from the Amvrakikos wetlands (e.g. [99, 100]), and now water buffalo populations have increased and spread to other wetlands (e.g. Spercheios).

It should be noted that even small-scale restoration works, unrelated to large, long-term projects, seem to be important for conservation education and awareness at the local level; this has been well demonstrated by the WWF Greece campaign for small wetlands on Crete and other islands [79]. Work on urban rivers is important as an "initiating enterprise" for restoration [101–104]. Interest in this aspect through the concept of green infrastructure in cities has recently been given increased attention [66, 105], but on-the-ground applications are scarce. Focus on riparian corridors in cities in Greece is a very new subject that should see important developments in the coming years [104].

New technologies for restoration in polluted waters, such as sewage treatment and water purification, are also on the rise [36], and some are of immediate interest for restoration of rivers and associated wetlands [106–108]. A major problem in Greece is olive oil mill treatment waste that is usually dumped directly into small rivers and streams; some pilot-scale work on this issue has begun [109].

Some evidence of notable conservation successes shows that strategic conservation actions can reduce the rate of species and population declines. The positive achievements of some of these species-centred conservation projects in Greece are seen primarily in effects on bird populations at specific sites. There is evidence that several species of birds requiring wetland and river delta habitats have increased during the last three decades [110]. In wetlands such as the Prespa, Kerkini, Karla, Evros, Amvrakikos and other sites, this is directly the result of targeted conservation and restoration efforts [111]. Since birds are not persecuted by humans as they were in the recent past, some populations have rebounded, such as pelican, cormorant and heron species [112]. Birds are good indicators of management of wetlands and other inland water bodies at the landscape scale [113, 114], and they have seriously promoted conservation designation and management [115] as well as restoration proposals in Greece (e.g. [116, 117]). Unfortunately, there are very few cases where there has been mention of benefits to other animals, such as fishes, from restoration projects in Greece (an exception is outlined in [118]). Despite the species-specific success stories, many species remain conservation dependent, requiring sustained, long-term investment that may require future restoration actions.

6 Failures, Shortcomings and Challenges

6.1 Problems with Past Restoration Projects

The largest restoration works that influence running water environments and their surroundings have very few follow-up actions, serious monitoring or adaptive management frameworks (e.g. Evrotas, Amvrakikos, Evros). In nearly all cases, the effectiveness and cost-effectiveness of these actions have been poorly assessed or monitored and disseminated in published works. One exception is the exemplary case of riparian woodland restoration in the Nestos river delta, which did provide a post-restoration monitoring plan [73], and in this case there is close cooperation with local Hellenic Forest Service workers and other stakeholders to monitor results at this important site.

It is rather unusual that most restoration efforts rest on large and often expensive engineering or forestry initiatives in high-profile degraded lentic or wetland habitats; there are very few efforts in the lotic environments and surprisingly few in small running waters and urban running water settings [45]. The lack of research, monitoring and understanding of river ecosystems functions, such as self-purification attributes of rivers, may also create complexities in applying ecological restoration [119]. Even some typical mitigation structures such as "fish ladders" in small hydroelectric works have no documented assessment of effectiveness in Greece.

Defining goals for the re-establishment of degraded river basin ecosystems such as drained foodplains and lakes has also created some examples of "failures." Of course, assessing the success of a project should always consider local sociocultural and political circumstances, yet guidelines for assessing the success of ecological restoration works are fairly simple (see [7]). Although not related to a lotic environment but a unique semi-isolated sub-basin of the Pinios river, the Lake Karla area was the most expensive restoration project in Greece [120] and was originally designed primarily as an ecological restoration action [88]. Instead of working to restore a natural lake-floodplain system, Lake Karla was created as a multifunctional and ambitious reservoir system with huge embankments and artificial water pumping works that would also supply future water for irrigation. This situation created some serious problems, including harmful algal blooms [120, 121]. Other problems where EU funds have been spent on reservoir or irrigation development instead of real ecological restoration actions have also been documented; two examples are the Atzan wetlands [122] and Lake Taka [123].

6.2 EU WFD Challenges

River restoration is a critical part of the EU WFD (Water Framework Directive 2000/60/EC), which is guided by river basin management plans (RBMPs) that should implement a programme of measures (PoMs) for degraded river water bodies. There has been considerable delay in the implementation of the Water Framework Directive actions in Greece, and this has already led the European Court of Justice to rule against Greece for not having completed the RBMPs on time [124]. The roots for the majority of the implementation problems are in governance and its changing architecture in Greece (see [125]). Also, since there is a poor history of limnological and ecosystem research in rivers, there is a poorly defined understanding of real restoration needs. Monitoring of conditions, especially the biota, began very recently at a nationwide scale [126]. However, despite these serious shortcomings, the framework for monitoring, planning and river basin management that the WFD has brought to Greece has seriously changed and challenged river management issues [127].

6.3 Current Management Difficulties

In many cases, restoring rivers and wetlands may be impossible, even if economic constraints could be surpassed (i.e. in urban and agricultural landscapes). Local socio-economics and politics influence the work's effectiveness by challenging the administration, planning and management related to these restoration projects; such coordination difficulties have been observed in many large-scale EU-funded projects in Greece [23]. In some cases, poor decision-making at the central and regional levels can lead to bad practices that resonate with negative aspects, e.g. the case of Lake Karla [120, 128] and difficulties with river conservation management at the Spercheios River [129]. In other cases, governmental control and development, even when guided by strict policy demands, may falter or fall behind, as is the case with Greece's river basin management plans [125, 126]. The economics and socio-economics of restoration planning are very important: perhaps in some cases it may be more effective to simply protect what you have than to focus on expensive narrow-scale restoration efforts. Such management difficulties become more complicated in times of economic and political instability. For 8 consecutive years, Greece has been greatly affected by the most severe economic crisis since WWII: a deep economic recession, sharp reductions in government spending and constant increase in unemployment rates. With this in mind, and despite the demands of EU and national policies, conservation and restoration actions may continue to fall further behind.

6.4 The Research-Policy Disconnect

EU Directives guide management objectives; however, optimal river basin management is hindered by a disconnect between policy-relevant research and action on the ground. Although the WFD is a stringent bureaucratic process, many aspects of its implementation in Greece seem to have somewhat progressed but with many difficulties. Some of the problems are grounded in ecosystem management concepts and applications [126]. One of the most important aspects is the importance of understanding aquatic ecosystem ecological integrity and designing restoration actions through the development of river-type-specific baselines and water body-specific targets. The WFD demands the use of type-specific reference conditions for assessment and as measures for restoration. This is correct in an ecological restoration framework, since the design of an ecological river restoration project should be based on a specified guiding image of a healthier river that could exist at the site [7]. However, the concept of "reference" is still debated within the scientific community and among management and conservation practitioners [6, 130]. Reference conditions can be historically based, geographically based or process based, and absolute or relative, depending on context and the specific spatio-temporal and ecosystemic thresholds. Understanding the natural history of varied river types is very important to restoration planning [131], and this is vital in any kind of ecosystem-based river management. There is increasing emphasis in Europe on river restoration driven by demands of the WFD; however, Greece has yet to promote restorative measures in its river water bodies [125]. Moreover, the current focus on instream aquatic conditions and aquatic biota that is routinely monitored ignores that many of the pronounced effects of degraded hydromorphology relate to the headwater intermittent streams, riparian zones, related wetlands and their wider floodplains. This problem is pronounced in the Mediterranean countries as a whole but is accentuated in data-scarce and poorly monitored situations such as the case of Greece [126].

Assessing the state of ecological integrity and measuring degradation are not easy or straightforward in Mediterranean river basins dominated by such long human history in complex cultural landscapes. The influence of humans on water resources is confounded by the region's inherent climate variability, since it drastically influences river flow regimes, river hydromorphology, biodiversity patterns and habitat structure. The degradation of the flow regime is the most widespread and often most destructive human-induced pressure in Mediterranean rivers; it is also difficult to accurately assess what is a natural or anthropogenic variation. Scientific reconstructions of the wider region's climate demonstrate a series of alternating periods with varying climatic characteristics with fluctuation lengths spanning from a few decades to many centuries [132]. In Crete, for example, during the little ice age (ca. 1,500–1,850 AD), it has been estimated that nearly two dozen perennial rivers run straight to the sea, whereas now less than five major streams do so during the summer months [12]. Part of this change is attributed to recent climate warming and part due to overexploitation for modern agriculture. Without in-depth study within each river basin, it is difficult to assess the degree that human-induced pressures have on recently altered waters [33, 40]. Building adequate and adaptive monitoring and multidisciplinary collaborative research at the regional and ecosystem scale is required for more effective and efficient restoration application in such complex conditions.

7 Unmet Needs and New Challenges

In the last few decades, three major types of restoration measures have been widely promoted in streams and rivers in Europe, America and Australia: riparian buffer and floodplain management, instream habitat enhancement and the removal of weirs and dams [131]. In Greece, even these basic actions have rarely been attempted in river environments, so there is much opportunity for this kind of policy-relevant restoration work in the future.

The following initiatives are deemed important in terms of ecological restoration, focusing particularly on river ecosystems and associated wetlands in Greece:

- Water pollution-related issues. Acute problems caused by poorly functioning sewerage treatment plants, small industry and poor controls on dumping still exist despite cleanup efforts.
- River floodplain restoration for flood control and flood protection. This synergy with restoration involves riverbed widening and embankment dismantling.
- Hydromorphological restoration in combination with habitat enhancement in protected areas. The channel, riparian zone, sediment and flow regimes may be restored in some cases through biodiversity conservation initiatives (e.g. in protected areas).
- Ecological flow issues below dams. This includes mitigation measures for fish migration (fish ladders).
- Water management issues in irrigation networks. Control and recharge, allowing more water to follow a natural flow regime in the extensive irrigated lowlands.
- Small urban stream restoration. Many cities could have greenways and rehabilitated stream reaches in an effort to expand constrained bed and riparian zones for green infrastructure and flood protection.
- Alien species management. Strategic management must be developed to clean problems associated with this form of "biological pollution."
- Monitoring and follow-up on restoration initiatives. These include works in the larger wetlands and river engineering areas; these can become "schooling" experiences for technique and educational development (Fig. 3).

Fig. 3 Challenges for restoration ecology in Greece (clockwise from *top left*): urban stream restoration, the flood-prone area of the Pikrodaphne stream in the Metropolitan Athens basin; road culvert creates artificial fish barrier in the Erymanthos river basin, Peloponnese; poorly designed urban spring-fed river park creation at the town of Skala, Peloponnese (Photos: S. Zogaris)

Some other outstanding and rather difficult issues that have intriguing idiosyncrasies and are important for biodiversity conservation in the future are outlined below:

7.1 Insect-Borne Diseases

After 2009 there is evidence that the malaria situation in Greece has been changing as locally acquired cases of *Plasmodium vivax* malaria have been repeatedly documented [133]. A reason for this spread is an influx of human immigrants from Asia and Africa and a change in the agricultural workforce that uses undocumented migrant workers, the majority of whom are from malaria-endemic tropical countries. As malaria cases and some other insect-borne diseases that are sometimes thought to be associated with "wetlands" increase, vector control will become important and may alter societal views of wetland and natural river habitats in the affected areas. The issue of malaria and other mosquito-borne diseases (such as West Nile virus) is serious for wetland and river conservation. Careful planning and multisectoral collaborations are needed both for human health protection and evidence-based treatment and for scientifically guided public awareness. Local communities and local government in Greece have recently wrongly targeted wetland drainage referring to mosquito control [29]. The issue of mosquito-borne disease and its relation to human migrants has recently been called a “public health tragedy” [134]. Great care is needed not to skew the public and policy approaches to wetland ecosystem conservation due this potentially serious health management issue.

7.2 Assisted Migration or Reintroduction of Fishes and Other Species Groups

Assisted migration refers to the human-aided translocation of select species or populations of plants and animals to suitable habitats outside their current ranges as well as to new sites within their current ranges. Although this issue is a source of debate among some conservationists [135], it is an imperative for saving rare species especially within conditions of extreme climate variability and change in situations of human-induced water stress and habitat and/or species population fragmentation [136, 137]. Already, some fish species have been extirpated and some are already extinct in the wild in Greece. This issue applies to fishes such as sturgeon, shad, salmonids, lampreys and local populations of several endemics in Greece. Many of these species are on the edge of extinction, and it seems that all sturgeon populations (four native species) have completely collapsed in Greece, in recent years [43]. Well-designed restocking programmes within a framework of fisheries management, protection measures and habitat rehabilitation may serve as valuable tools for reintroduction or enhancement of wild stocks. However, the risk of losing genetic variability, which happened after the massive restocking programmes abroad, should be thoroughly considered in advance. A strong level of scientifically led conservation genetics is required to do this kind of work (see [138]). Unfortunately, most previous efforts in Greece have had poor or no results (e.g. [139, 140]).

7.3 Managing “Novel Aquatic Ecosystems”

Heavily modified and artificial water systems may drastically change ecological integrity of wider river basins. Artificial water bodies, such as ditches and artificial channels, trans-basin transfer canals and instream reservoirs have been constructed in many river basins in Greece, and these are rarely managed for their biodiversity or ecological potential [83, 141]. Recently, these human-modified systems are

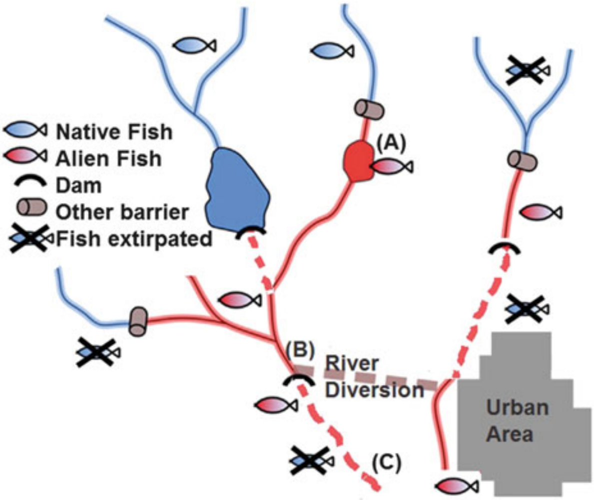


Fig. 4 Schematic representation of the interactive effects of artificial changes in aquatic connectivity and intermittency producing significant biotic changes and novel freshwater ecosystems. (**a**) Alien fish stocking in a lake spreads invasive fish species (*red* stretches), and (**b**) inter-basin diversion of stream flow to a city in a neighbouring basin (indicated by broken *red/grey line*) further assists invasive spread. Native fish species populations become fragmented into a series of isolated populations in headwaters (*blue-shaded* streams) above artificial barriers. (**c**) Artificial barriers and dams have produced stretches where fish are extirpated, and some of these are artificially intermittent (shown by *dotted line* for the river stretch) Adapted from [144]

being called novel aquatic systems, and their management is being treated as being important for biodiversity [142, 143]. The interaction of these systems with natural stream conditions and the spread of alien species is a growing conservation concern [144], and often several basins may be degraded when human-induced connectivity, artificial flow and alien species interact (Fig. 4). The degraded novel systems may function as reservoirs for the spread of alien invasive species further degrading ecological integrity. A scientifically led precautionary approach is needed to assess both the negative impacts and conservation opportunities provided by novel aquatic ecosystems.

7.4 Adaptations to Climate Change

A key characteristic in Mediterranean climate conditions is the remarkable climatic variability observed during the last few centuries [132]. The southern Balkans are among the regions which are predicted to become drier under IPCC climate scenarios, and the hydrological effects have concerned several researchers [145]. In recent years, there has been evidence of increased climatic dryness in parts of Greece, and an alarming indicator is the decline of spring-fed water systems [146]. Hydromorphological and flow regime change evidence is also being compiled (e.g. [147]). Ecohydrological modelling has shown that predicted scenarios will alter ecosystems, for example, cold-water biota such as trout streams [148]. For the conservation of biodiversity, it is important to focus on four initiatives for adaptation to climate change: (a) land and water protection and management, (b) direct species management, (c) monitoring and planning, and (d) law and policy (see [149]). Gaining experience in restoration and in assessing and understanding scientific ecosystem change and evolution is critical for effective adaptation to this broad-scale and largely poorly predictable environmental change. The need to promote adaptive management frameworks in aspects of conservation and monitoring is also an imperative within the climate change context.

8 Conclusions

Until recently a wholly anthropocentric development worldview has exploited rivers and wetlands solely as commodities and for human health risks (as related to insect-borne disease and flooding). This paradigm has changed in Greece as many rivers and wetlands have been designated as protected areas primarily for their biodiversity. Still, progress in integrated conservation and restoration has been very slow [51]. There are many opportunities to develop restoration in rivers in Greece; some important approaches include the following:

– The WFD's programmes of measures (PoMs) for water bodies represent the most important opportunity for widespread policy-relevant restoration in river corridors.
– Ecological flow measures are an important unmet challenge, and beneath many hydroelectric dams, there are significant degradations due to flow regime alteration (e.g. hydropeaking). Holistic approaches should be developed that are both policy relevant and satisfy site-based optimal mitigation measures.
– Reintroduction or assisted migration schemes for fishes and other species could assist both biodiversity conservation and community restoration; fish pass construction is also important and poorly implemented in Greece.
– Biodiversity restoration applications should work synergistically with WFD demands in protected areas (e.g. aimed specifically at habitats and species assemblages that have been degraded by hydrological and hydromorphological changes).
– Antipollution initiatives and strictly enforced regulations can make a very big difference especially targeting point-source pollution problems (light industry, agriculture, sewage treatment plant outfalls).
– Taking advantage of land abandonment, which facilitates renaturing and rewilding in river riparian zones, could promote the conservation/restoration of floodplain buffers.

– Riparian and river restoration in urban and peri-urban areas in combination with flood protection schemes should enhance best practice, green infrastructure and public awareness.
– Involving the public in volunteer and citizen sciences, that is, promoting restoration actions also for education, recreation and ecotourism. This is again important especially in urban, peri-urban and touristic protected areas.

Significant positive synergies may be created especially with respect to flood control and river engineering requirements in agricultural areas and WFD measures within protected areas. An example of this is "river widening" engineering practices, since this can become an important tool to link ecological objectives with flood protection and habitat enhancement, recreating multichannel networks in previously artificially constrained channelised systems. Although this is now widely practised in Western Europe, efforts in Greece are usually only at the proposal stage (e.g. [101]).

Ecological restoration is not an easy and straightforward undertaking [8], especially in Mediterranean rivers [150]. In Greece, a problem is the disconnect among scientists, society and conservation/water management practitioners. Since the 2008 economic crisis, it is inevitable that many members of the public may see restoration actions as a luxury. This negative perspective must change for serious broad-scale restoration work to move forward. It has been shown that volunteer involvement is extremely important and valuable for guiding conservation planning and promoting positive stakeholder involvement and science-guided public awareness. Citizen science may also provide screening-level information for river and wetland conditions; data from participatory monitoring networks are not less informative and may sometimes be more informative, than those collected in professional schemes [151]. The Natura 2000 protected sites and their management agencies could play a leading role in providing best practice applications that involve citizen scientists [64]. Scientific monitoring of rivers is now a policy-relevant imperative, and this should develop into an adaptive monitoring approach that can guide, prioritise and better inform conservation and restoration needs.

References

1. Bolling DM (1994) How to save a river. Island Press, Washington, DC
2. Naiman RJ, Décamps H, McClain ME (2005) Riparia: ecology, conservation, and management of streamside communities. Elsevier/Academic Press, San Diego
3. Ormerod SJ (2004) A golden age of river restoration science? Aquat Conserv 14:543–549
4. SER (2004) The SER international primer on ecological restoration. Society for Ecological Restoration International Science and Policy Working Group. www.ser.org Tucson, USA, 13 pp
5. Zalidis GC, Crisman TL, Gerakis PA (2002) Restoration of Mediterranean wetlands. Hellenic Ministry of the Environment, Physical Planning and Public Works, Athens and Greek Biotope/Wetland Centre, Thermi

6. Morandi B, Piégay H, Lamouroux N, Vaudor L (2014) How is success or failure in river restoration projects evaluated? Feedback from French restoration projects. J Environ Manage 137:178–188
7. Palmer MA, Bernhardt ES, Allan JD, Lake PS, Alexander G, Brooks S, Carr J, Clayton S, Dahm CN, Follstad Shah J, Galat DL (2005) Standards for ecologically successful river restoration. J Appl Ecol 42:208–217
8. Hilderbrand RH, Watts AC, Randle AM (2005) The myths of restoration ecology. Ecol Soc 10:19. [Online]. http://www.ecologyandsociety.org/vol10/iss1/art19/
9. Angelakis AN, Dialynas MG, Despotakis V (2012) Evolution of water supply technologies in Crete, Greece through the centuries. In: Evolution of water supply throughout millennia. IWA Publishing, London, UK, Ch 9. pp 227–258
10. Papadimos DL (1974) The hydraulic works of the ancients. Ekdosis Technikou Epimelitiriou tis Ellados, Athens. Vols. B and C. (In Greek)
11. Koutsoyiannis D, Angelakis AN (2004) Agricultural hydraulic works in ancient Greece. In: Stewart BA, Howell T (eds) The encyclopedia of water science. Marcel Dekker, New York, pp. 415–417
12. Rackham O, Moody J (1996) The making of the Cretan landscape. Manchester University Press, Manchester
13. Kouvelis S (1996) Water resources. In: Papaspiliopoulos S, Papagiannis TH (eds) The environment in Greece 1991-1996. Idrima Bodosaki, Athens, pp 108–122 (in Greek)
14. Maragou P, Mantziou D (2000). Assessment of the Greek Ramsar wetlands. WWF-Greece, pp 59 + Answered questionnaires
15. Abell R, Thieme ML, Revenga C, Bryer M, Kottelat M, Bogutskaya N, Coad B, Mandrak N, Balderas SC, Bussing W, Stiassny ML (2008) Freshwater ecoregions of the world: a new map of biogeographic units for freshwater biodiversity conservation. Bioscience 58(5):403–414
16. Catsadorakis G, Paragamian K (2007) Inventory of the wetlands of the Aegean Islands: identity, ecological status and threats. WWF Greece, Athens (in Greek)
17. Zogaris S, Skoulikidis N, Economou AN, Bobori D, Ghini M, Stergiou KI (eds) (2016) EU directive 2000/60 and the conservation of inland waters: research and prospects. IMBRIW, Hellenic Centre for Marine Research, Athens (In Greek)
18. Soulé ME (1986) Conservation biology: the science of scarcity and diversity. Sinauer Associates, Sunderland
19. Chatzinikolaou Y (2016) National monitoring programme: ecological status and biological quality elements. In: Zogaris S, Skoulikidis N, Economou AN, Bobori D, Ghini M, Stergiou KI (eds) EU directive 2000/60 and the conservation of inland waters: research and prospects. IMBRIW Hellenic Centre for Marine Research. Athens, pp 11–13 (in Greek)
20. Skoulikidis N (2016) Presentation of the monitoring programme. In Zogaris S, Skoulikidis N, Economou AN, Bobori D, Ghini M, Stergiou KI (eds) EU directive 2000/60 and the conservation of inland waters: research and prospects. IMBRIW, Hellenic Centre for Marine Research, Athens, pp 7–10. (In Greek)
21. Moser M (2000) Wetland status and trends in Europe: the case for rehabilitation and restoration of naturally functioning wetlands. In: Implementing the EU water framework directive seminar, vol 2, pp 29–36
22. Psilovikos A (1992) Changes in Greek wetlands during the twentieth century: the cases of the macedonian inland waters and of river deltas of the Aegean and Ionian coasts. In: Conservation and management of Greek wetlands – proceedings of a Greek Wetlands Workshop, held in Thessaloniki, Greece, IUCN, pp 175–196
23. PAF (Prioritised Action Framework) (2014) Report setting priorities for nature conservation within the Natura 2000 network in Greece (council directive 92/43/EEC, directive 2009/147/EC). Department of Biodiversity and Protected Areas of the Directorate for Biodiversity Protection, Soil and Waste Management, Ministry of Environment, Energy and Climate Change

24. Danis K, Lenglet A, Tseroni M, Baka A, Tsiodras S, Bonovas S (2013) Malaria in Greece: historical and current reflections on a re-emerging vector borne disease. Travel Med Infect Dis 11:8–14
25. Pantelouris EM (1980) Greece: an introduction. Blueacre Books, Glasgow, Scotland
26. Tsiouris SE, Gerakis PA (1991) Wetlands of Greece: values, alterations, conservation. WWF, Laboratory of Ecology and Environmental Protection (Faculty of Agriculture, Aristotelian University of Thessaloniki). (In Greek)
27. Goutner V, Jerrentrup H (1987) The destruction of Drana lagoon in the Evros Delta Ramsar Wetland, and its consequences to waterfowl. Wader Study Group Bull 50:18–19. 47
28. Zogaris S, Chatzinikolaou Y, Dimopoulos P (2009) Assessing environmental degradation of montane riparian zones in Greece. J Environ Biol 30:719–726
29. Zogaris S, Konstantinis A, Giakoumi S (2011) Wetland drainage at Acheron Delta: it's not only the Acheron Goby that is threatened. I Physi (Bull Hellenic Soc Protect Nat) 134:17–20. (In Greek)
30. Korakis G, Gerasimidis AC (2006) Coastal and halophytic habitats and their flora in Evrotas Delta (SE Peloponnisos, Greece). Biol Res 6:155–166
31. Skoulikidis N, Economou AN, Gritzalis KC, Zogaris S (2009) Rivers of the Balkans, In: Tockner K, Uehlinger U, Robinson CT (eds) Rivers of Europe. Elsevier Academic Press, Amsterdam, pp 421–466
32. Tsouni A, Zervos N, Hadjibiros K, Adreadakis A (2002) An environmental database for the status of freshwater in Greece. Global Nest 4:1–14
33. Skoulikidis NT (2009) The environmental state of rivers in the Balkans – a review within the DPSIR framework. Sci Total Environ 407:2501–2516
34. Gerakis A, Kalburtji K (1998) Agricultural activities affecting the functions and values of Ramsar wetland sites of Greece. Agric Ecosyst Environ 70:119–128
35. Konstantinou IK, Hela DG, Albanis T (2006) The status of pesticide pollution in surface waters (rivers and lakes) of Greece. Part I. Review on occurrence and levels. Environ Pollut 141:555–570
36. Tsihrintzis VA, Gikas GD (2010) Constructed wetlands for wastewater and activated sludge treatment in North Greece: a review. Water Sci Technol 61:2653–2672
37. Zalidis G (1998) Management of river water for irrigation to mitigate soil salinization on a coastal wetland. J Environ Manage 54:161–167
38. Koutrakis ET, Emfietzis G, Sylaios G, Zoidou M, Katsiapi M, Moustaka-Gouni M (2016) Fish kill in Ismarida Lake, Greece: identification of drivers contributing to the event. Mediterr Mar Sci 17:280–291
39. Patsia A, Vafeiadou A, Michalopoulou A, Samartzi S, Mallinis D, Bobori DC (2016) Fish massive kills in Bogdanas stream (Mygdonia basin). Proc Hellenic Conf Ichthyologists 16:305–305
40. Skoulikidis NT, Vardakas L, Karaouzas I, Economou AN, Dimitriou E, Zogaris S (2011) Assessing water stress in a Mediterranean lotic system: insights from an artificially intermittent river in Greece. Aquat Sci 73:581–597
41. Legakis A, Maragou P (eds) (2009) Red data book of threatened animal species of Greece. Hellenic Zoological Society, Athens
42. Economou AN, Barbieri R, Daoulas Ch, Psarras T, Stoumboudi M, Bertahas I, Giakoumi S, Patsias A (1999) Endangered freshwater fish of western Greece and the Peloponnese – distribution, abundance, threats and measures for protection. Final technical report, National Centre for Marine Research, Athens (in Greek)
43. Barbieri R, Zogaris S, Kalogianni E, Stoumboudi M, Chatzinikolaou Y, Giakoumi S, Kapakos Y, Kommatas D, Koutsikos N, Tachos V, Vardakas L, Economou AN (2015) Freshwater fishes and lampreys of Greece: an annotated checklist. Monographs on Marine Sciences No. 8. Hellenic Center for Marine Research: Athens

44. Papadaki C, Soulis K, Muñoz-Mas R, Martinez Capel F, Zogaris S, Ntoanidis L, Dimitriou E (2016) Potential impacts of climate change on flow regime and fish habitat in mountain rivers of the southwestern Balkans. Sci Total Environ 540:418–428
45. Zaimes GN, Gounaridis D, Iakovoglou V, Emmanouloudis D (2011) Riparian area studies in Greece: a literature review. Fresen Environ Bull 20:1470–1477
46. Chatzinikolaou Y, Ntemiri K, Zogaris S (2011) River riparian zone assessment using a rapid site-based index in Greece. Fresen Environ Bull 20:296–302
47. Zenetos A, Pancucci-Papadopoulou MA, Zogaris S, Papastergiadou E, Vardakas L, Aligizaki K, Economou AN (2009) Aquatic alien species in Greece (2009): tracking sources, patterns and effects on the ecosystem. J Biol Res Thessalon 12:135–172
48. Muhlfeld CC, Kalinowski ST, McMahon TE, Taper ML, Painter S, Leary RF, Allendorf FW (2009). Hybridization rapidly reduces fitness of a native trout in the wild. Biol Lett. doi: 10.1098/rsbl.2009.0033
49. Papageorgiou K, Vogiatzakis IN (2006) Nature protection in Greece: an appraisal of the factors shaping integrative conservation and policy effectiveness. Environ Sci Policy 9:476–486
50. Lekakis JN, Kousis M (2013) Economic crisis, Troika and the environment in Greece. S Eur Soc Polit 18:305–331
51. Apostolopoulou E, Pantis JD (2009) Conceptual gaps in the national strategy for the implementation of the European Natura 2000 conservation policy in Greece. Biol Conserv 142:221–237
52. Keulartz J (2009) European nature conservation and restoration policy-problems and perspectives. Restor Ecol 17:446–450
53. Pavlikakis GE, Tsihrintzis VA (2006) Perceptions and preferences of the local population in Eastern Macedonia and Thrace National Park in Greece. Landscape Urban Plann 77:1–16
54. Christopoulou I, Roumeliotou V (2006) Uniting people through nature in Southeast Europe: the role (and limits) of nongovernmental organizations in the transboundary Prespa Park. Southeast Eur Black Sea Stud 6:335–354
55. Hoffmann L, Olney PJ, Swift J (1964) IUCN/IWRB expedition January–February 1964. IWRB Report, Slimbridge, England
56. Weber N, Christophersen T (2002) The influence of non-governmental organisations on the creation of Natura 2000 during the European policy process. Forest Policy Econ 4:1–12
57. Fourniotis NT (2012) A proposal for impact evaluation of the diversion of the Acheloos River, on the Acheloos estuary in western Greece. Int J Eng Sci Tech 4:1792–1802
58. Gerakis PA, Koutrakis ET (eds) (1996) Greek wetlands. Greek Biotopes-Wetland Centre – EKBY. Thermi (in Greek)
59. Koussouris T (1998) The water in nature, in development, in environmental protection. Monographs on Marine Sciences No. 1, National Centre for Marine Research (in Greek)
60. Christopoulou OG, Tsachalidis E (2004) Conservation policies for protected areas (wetlands) in Greece: a survey of local residents' attitude. Water Air Soil Pollut Focus 4:445–457
61. Crisman TL, Takavakoglou V, Alexandridis T, Antonopoulos V, Zalidis G (2009) Rehabilitation of abandoned saltworks to maximize conservation, ecotourism and water treatment potential. Global NEST J 11:24–31
62. Karavellas D, Catsadorakis G, Maragou P, Nantsou T, Svoronou E (2003) Management of protected areas: best practice guide. WWF – Greece, Hellenic Ministry of Environment, Land Planning and Public Use
63. Hellenic Ministry of Environment, Energy and Climate Change (2014) National Biodiversity Strategy and Action Plan (P.D. Dimopoulos, Scientific Coordinator), Athens
64. Sylaios G (ed) (2006) The management of water, flora and fauna in Greek wetlands: experiences from life-nature III projects: networking report. Prefectural Authority of Drama-Kavala-Xanthi. (In Greek, with English Summary)
65. Orphanidis C (ed) (1987) The ecological movement in Greece. Meta Tin Vrochi Press. (In Greek)

66. Belavilas N, Vatavili F (2009) Greenspace and open spaces in cities. Series: Guide for the Environment. WWF Greece (in Greek)
67. Valaora G, Dimalexis T (2007) Implementation of LIFE nature projects in Greece. In: Primack R, Diamantopoulos J, Arianoutsou M, Daniilidis D, Valakos S, Pafilis P, Pantis JD (eds) Conservation and protection of biodiversity. Crete (Greece): AEI Publications, pp 387–398
68. Theocharis M, Arapis T, Zogaris S (2004a) Engineering the restoration of a coastal freshwater wetland: a sluice-and-canal scheme in Rodia Swamp, Amvrakikos, Greece. International conference on agricultural engineering AgEng 2004 "Engineering the Future", 12–16 September 2004, Leuven, Belgium. Conference proceedings, vol II, pp 804–805
69. Theocharis M, Zogaris S, Economou AN, Kapsimalis V, Dimopoulos P (2004b) Restoration actions and monitoring at a Mediterranean river floodplain wetland: the Amvrakikos case-study. V International symposium on ecohydraulics "Aquatic Habitats: Analysis and Restoration", 11–17 Madrid, Spain. IAHR congress proceedings, vol I, pp 582–588
70. Zogaris S, Kardakari N, Rigas I, Arapis Th (2006) Reedbed restoration at Amvrakikos, Greece. In: White G, Purps J, Alsbury S (eds) The bittern in Europe: a guide to species and habitat management. RSPB, LIPO, The Brandenburg State Office of the Environment, pp 131–133 & Appendix I, p 173
71. Zogaris S, Hatzirvassanis V, Loi I, Vlamis-Gardikas A (2008) Several landowners in a protected area: riparian woodland at Amvrakikos. In: Arizpe D, Mendes A, Rabaca J (eds) Sustainable riparian zones: a management guide. Generalitat Valenciana, Valencia, Spain, pp 251–267
72. Kakouros P, Katsakiori M (2012) Integrated actions for the restoration and interpretation of the riparian forest of Nestos, Greece. Conference proceedings, protection and restoration of the environment XI, pp 1120–1128
73. Kakouros P, Dafis S (2010) Monitoring program of the vegetation restoration works of the riparian forest of Nestos, 2nd edn. Greek Biotope-Wetland Centre, Thermi
74. Efthimiou G, Smiris P (2002) Management, Protection and restoration measures of the riparian ecosystems of Nestos Delta. In: Proceedings of the international conference "protection and restoration of the environment VI" Skiathos, Greece, vol III, pp 1871–1874
75. Efthimiou G, Jerrentrup H (2013) Results of riparian forest restoration in Nestos Delta, NE-Greece 10 years after plantation. European River restoration conference, 11–13 Sept. Vienna, Poster Presentation
76. Aperghis GG, Gaethlich M (2006) The natural environment of Greece: an invaluable asset being destroyed. Southeast Eur Black Sea Stud 6:377–390
77. Tzali M, Fric J, Promponas N (2009). The birds of wetlands in Attiki. Bird monitoring programme on wetlands of Attiki. Hellenic Ornithological Society, Athens (In Greek)
78. Zogaris S, Karaouzas I, Kastritis Th, Kaniastas E, Stavakas L, Trigou R, Vlami V (2015) Artemis wetland (Artemis, Attika, Greece): biodiversity conservation study. Hellenic Ornithological Society/Hellenic Centre for Marine Research - Institute of Marine Biological Resources and Inland Waters, Athens (in Greek)
79. Giannakakis T, Paragamian K (2015) Island wetlands: an outline of the conservation of Greece's island wetlands initiative (factsheet). http://www.wwf.gr/images/pdfs/IslandWetlandsFSint.pdf
80. Tsihrintzis VA, Kotzageorgis G, Niadas Y, Papagrigoriou S (2003) Modeling the restoration of the Schinias wetland related to the 2004 Olympic Games Rowing Center construction. HELECO '03 Environmental Conference and Exposition, Technical Chamber of Greece, Marousi, Athens, Greece, vol C, pp 263–272 (in Greek)
81. Zalidis GC, Mantzavelas AL (eds) (1994) Inventory of Greek wetlands as natural resources. EKBY.Biotope/Wetland Centre (EKBY). Thermi, Greece
82. Gerakis PA, Tsiouris S, Tsiaoussi V (2007) Water regime and biota: proposed minimum values of lakes water level and of rivers discharge of Macedonia and Thrace, Greece. Goulandris Natural History Museum/Greek Biotope-Wetland Centre, Thermi (in Greek)

83. Georgiadis NM, Paragamian K, Giannakakis T, Poursanidis D, Catsadorakis G (2010) Types of artificial water bodies in the Aegean islands (Greece), their environmental impact and potential value for biodiversity. In: Christodoulou, Stamou (eds) Environmental hydraulics. Taylor and Francis Group, London
84. Dimitriou E, Mentzafou A, Zogaris S, Tzortziou M, Gritzalis K, Karaouzas I, Nikolaidis C (2012) Assessing the environmental status and identifying the dominant pressures of a transboundary river catchment, to facilitate efficient management and mitigation practices. Environ Earth Sci 66:1839–1852
85. Dimalexis A, Spyridaki E, Dretakis M, Nikolakakis M (2005) Good practice guide for the design and management of reservoirs in order to achieve multiple functions. Life 00 ENV/GR/000685. Natural History Museum of Crete. University of Crete. Heraklion (In Greek)
86. Mitraki C, Crisman TL, Zalidis G (2004) Lake Koronia, Greece: shift from autotrophy to heterotrophy with cultural eutrophication and progressive water-level reduction. Limnologica 34:110–116
87. Zalidis GC, Takavakoglou V, Alexandridis T (2004a) Revised restoration plan of Lake Koronia. Unpublished report. Aristotle University of Thessaloniki, Department of Agronomy, Laboratory of Applied Soil Science. (In Greek, English summary)
88. Zalidis GC, Takavakoglou V, Panoras A, Bilas G, Katsavouni S (2004b) Re-establishing a sustainable wetland at former Lake Karla, Greece, using Ramsar restoration guidelines. Environ Manag 34:875–886
89. Loukas A, Mylopoulos N, Vasiliades L (2007) A modeling system for the evaluation of water resources management strategies in Thessaly, Greece. Water Resour Manag 21:1673–1702
90. Chatziapostolou A, Siavalas G, Kalaitzidis S, Christanis K (2013) Geological Study for a wetland restoration: the case of the drained Mouria Lake (W. Peloponnese). Bulletin of the Geological Society of Greece. 2013 Proceedings of the 13th international congress, Chania, vol XLVII Sept 2013 XLVII, No 1–72
91. Ganatsas P, Tsakaldimi M, Katsaros D (2013) Natural resource management in national parks: a management assessment of a Natura 2000 wetlands site in Kotychi-Strofylia, southern Greece. Int J Sust Dev World 20:152–165
92. Valaoras G, Dimalexis T (2006) Implementation of life-nature projects in Greece, a preliminary evaluation 3rd conference of the Hellenic Ecological Society and Hellenic Zoological Society, Ioannina
93. Daoulas Ch (ed) (1998) Actions for the conservation and restoration of the Greek Stickleback, a threatened endemic fish. Ministry of Environment/National Centre for Marine Research (in Greek)
94. Jones W, Eldridge J, Silva JP, Schiessler N (2007) LIFE and Europe's rivers: protecting and improving our water resources. European Communities, Luxemburg
95. Barbieri R, Stoumboudi MTh, Kalogianni E, Corsini-Foka M (2003) Abundance evaluation of fish populations – the case of gizani (*Ladigesocypris ghigii*). Proceedings of 7th Hellenic symposium on oceanography and fisheries, Crete
96. Silva JP, Toland J, Nottingham S, Jones W, Eldridge J, Hudson T, Heppner K, McGlynn D, Thevignot C (2015) LIFE and freshwater fish. Brussels
97. Kazoglou Y (2009) Meadow management in Greece. Report from the EU LIFE Nature Seminar on Restoration of Meadow Bird Habitats, LIFE-Nature, Copenhagen
98. Kazoglou Y, Papanastasis VP (2003) Effects of water buffalo (*Bubalus bubalis*) grazing on the vegetation of the littoral zone of Lake Mikri Prespa. In: Platis PD, Papachristou TG (eds) Range science and development of mountainous regions. Proceedings of the 3rd Panhellenic rangeland congress in Karpenissi, 4–6 September 2002. Hellenic Pasture and Range Society (in Greek with English summary)
99. Riddell ES (2002) The effects of water buffalo (*Bubalus bubalis*) on wetland bird habitat: implications for habitat management in the Amvrakikos wetlands, Greece. Magister in Scientia in Ecology, Dissertation, School of Biological Sciences, University of Wales, Bangor

100. Zogaris S, Kazoglou Y, Rigas I, Vrettou E (2000) Pilot management actions in reed-swamp through the re-introduction of Water Buffalo (*Bubalus bubalis*) at the Amvrakikos Gulf. Life-Nature 99 "Conservation Management of Amvrakikos Wetlands". Life99NAT/GR/006475). Oikos Ltd./Development Agency of Amvrakikos (ETANAM S.A) (In Greek)
101. Dimitriou E, Markogianni V, Mentzafou A, Karaouzas I, Zogaris S (2014) Ecological status assessment of Pikrodafni stream (Attica, Greece), restoration and management measures. Desalin Water Treat 56:1248–1255
102. Kantartzis A, Varras G, Kakouri P, Koutsikou M, Papadopoulou A, Gogolou C (2006) Greenway planning and management of Urban riparian corridors: the alternative basis for an integrated system of Urban Green Spaces. Case study: riparian corridors in the city of Igoumenitsa, Greece', proceedings of the 5th WSEAS international conference on environment, ecosystems and development, Venice, Italy, pp 168–174
103. Xanthouli S, Tsihrintzis VA (2003) Creation of a multiple-use riparian corridor in Chryssoupolis, Greece, HELECO '03, environmental conference and exposition, technical chamber of Greece, Marousi, Athens, Greece, vol C, pp 273–280
104. Iakovoglou V, Zaimes GN, Gounaridis D (2013) Riparian areas in urban settings: two case studies from Greece. Int J Innov Sustain Dev 7(3):271–288
105. Tzoulas K, Korpela K, Venn S, Ylipelkonen V, Kazmierczak A, Niemela J, James P (2007) Promoting ecosystem and human health in urban areas using green infrastructure: a literature review. Landscape Urban Plan 81:167–178
106. Akratos CS, Tsihrintzis VA, Pechlivanidis I, Sylaios GK, Jerrentrup H (2006) A free-water surface constructed wetland for the treatment of agricultural drainage entering Vassova Lagoon, Kavala, Greece. Fresen Environ Bull 15:1553–1562
107. Akratos CS, Tsihrintzis VA (2004) Pilot-scale constructed wetlands for wastewater treatment studies, proceedings of the international conference on protection and restoration of the environment VII, 28 June–1 July 2004, Mykonos, Greece
108. Haralambidou KI, Tsihrintzis VA, Sylaios GK (2003) Control of saline wedge intrusion in the estuary of Strymonas River using an air curtain, proceedings of the 8th conference on environmental science and technology, September 8–10, 2003, Porto Myrina Palace, Lemnos, Greece, vol A, pp 302–309
109. Herouvim E, Akratos CS, Tekerlekopoulou A, Vayenas DV (2011) Treatment of olive mill wastewater in pilot-scale vertical flow constructed wetlands. Ecol Eng 37:931–939
110. Handrinos G, Kazantzidis S, Alivizatos Ch, Akriotis T, Portolou D (2015) International waterbird census in Greece (1968–2006). Hellenic Ornithological Society – Hellenic Bird Ringing Centre, Athens
111. Maragou P (2000) Conservation of *Phalacrocorax pygmeus* and *Anser erythropus* in Greece. Summary technical review on non-recurring biotope management actions. Life-Nature II Programme, WWF-Greece, HOS, SPP. (In Greek)
112. Catsadorakis G, Onmuş O, Bugariu S, Gül O, Hatzilacou D, Hatzofe O, Malakou M, Michev T, Naziridis T, Nikolaou H, Rudenko A (2015) Current status of the Dalmatian pelican and the great white pelican populations of the Black Sea/Mediterranean flyway. Endanger Species Res 27:119–130
113. Ortega-Álvarez R, Lindig-Cisneros R (2012) Feathering the scene: the effects of ecological restoration on birds and the role birds play in evaluating restoration outcomes. Ecol Restor 30 (2):116–127
114. Platteeuw M, Foppen RPB, van Eerden MR (2010) The need for future wetland bird studies: scales of habitat use as input for ecological restoration and spatial water management. Ardea 98:403–416
115. Hoffmann L, Hafner H, Salathé T (1996) The contribution of colonial waterbird research to wetland conservation in the Mediterranean region. Colon Waterbirds 19:12–30
116. Goutner V (1997) Use of the Drana Lagoon (Evros Delta, Greece) by threatened colonially nesting waterbirds and its possible restoration. Biol Conserv 81:113–120

117. Vougioukalou M, Akriotis T, Dimalexis A (2011) Waterfowl habitat restoration in a seasonal island wetland. Procedia Environ Sci 9:191–195
118. Koutrakis E, Sylaios G, Kamidis N, Markou D, Sapounidis A (2009) Fish fauna recovery in a newly re-flooded Mediterranean coastal lagoon. Estuar Coast Shelf Sci 83:505–515
119. Ioannou A, Chatzinikolaou Y, Lazaridou M (2009) A preliminary pressure-impact analysis applied in the Pinios river basin (Thessaly, Central Greece). Water Environ J 23:200–209
120. Sidiropoulos P, Papadimitriou T, Stabouli Z, Loukas A, Mylopoulos N, Kagalou I (2012) Past, present and future concepts for conservation of the re-constructed Lake Karla (Thessaly-Greece). Fresen Environ Bull 21:3027–3034
121. Gialis S, Laspidou CS (2014) Lake Karla and the contradictory character of Greek environmental policies: a brief historical overview. In: IWA-regional symposium on water, wastewater and environment, pp 22–24
122. Bousbouras, D. (2009) Action plan for the special protection area "GR1230005 Perichi Elous Artzan". In: Dimalexis A, Bousbouras D, Kastritis T, Manolopoulos A, Saravia V (eds) Final project report for the evaluation of 69 important bird areas as special protection areas. Hellenic Ministry for the Environment, Physical Planning and Public Works, Athens (in Greek)
123. Bousbouras D, Dimalexis A (2005) Lake Taka, a victim of bad management. Oionos Mag. Hellenic Ornithological Society 23:14–15 (In Greek)
124. Nantsou T (2014) Environmental legislation in Greece – 10th annual review – summary. WWF Greece, Athens
125. Kanakoudis V, Tsitsifli S (2015) River basin management plans development in Greece based on the WFD 2000/60/EC guidelines. Desalin Water Treat 56:1231–1239
126. Economou AN, Zogaris S, Vardakas L, Koutsikos N, Chatzinikolaou Y, Kommatas D, Kapakos Y, Giakoumi S, Oikonomou E, Tachos V (2016) Developing policy-relevant river fish monitoring in Greece: insights from a nation-wide survey. Mediterr Mar Sci 171:302–322
127. Kanakoudis V, Tsitsifli S, Azariadi T (2015) Overview of the river basin management plans developed in Greece under the context of the water framework directive 2000/60/EC focusing on the economic analysis. Water Resour Manag 29:3149–3174
128. Papadimitriou T, Katsiapi M, Moustaka-Gouni M, Kagalou I (2013) Artificially-born "killer" lake: phytoplankton based water quality and microcystin affected fish in a reconstructed lake. Sci Total Environ 452–453:116–124
129. Oikonomou EN, Dimitriou E, Papadopoulos T, Tzitzis AL, Politi N, Kalaitzakis N, Zogaris S, Giakoumi S, Economou AN (2015) Technical report concerning stakeholders questionnaire results and conclusions with respect to the river basin management vision. Kripis Project-Sperchios. Institute of Marine Biological Resources and Inland Waters, HCMR, Athens (in Greek)
130. Woodward G, Friberg N, Hildrew AG (2010) Science and non-science in the biomonitoring and conservation of fresh waters. In: de Carlo F, Bassano A (eds) Freshwater ecosystems and aquaculture research. Nova Science Publishing, pp 277–288
131. Feld CK, Birk S, Bradley DC, Hering D, Kail J, Marzin A, Melcher A, Nemitz D, Pedersen ML, Pletterbauer F, Pont D, Verdonschot PFM, Friberg N (2011) From natural to degraded rivers and back again: a test of restoration ecology theory and practice. Adv Ecol Res 44 (3):119–209
132. Markonis Y, Batelis SC, Dimakos Y, Moschou E, Koutsoyiannis D (2016b) Temporal and spatial variability of rainfall over Greece. Theor Appl Climatol:1–16
133. Sudre B, Rossi M, Van Bortel W, Danis K, Baka A, Vakalis N (2013) Mapping environmental suitability for malaria transmission. Greece Emerg Infect Dis [Internet]. doi:10.3201/eid1905.120811
134. Bonovas S, Nikolopoulos G (2012) High-burden epidemics in Greece in the era of economic crisis. Early signs of a public health tragedy. J Prev Med Hyg 53:169–171

135. McLachlan JS, Hellmann JJ, Schwartz MW (2007) A framework for debate of assisted migration in an era of climate change. Conserv Biol 21:297–302
136. Keenleyside KA, Dudley N, Cairns S, Hall CM, Stolton S (2012) Ecological restoration for protected areas: principles, guidelines and best practices. IUCN, Gland
137. SER (2009) Ecological restoration and rare species management in response to climate change. Pol Position Statement, 7 pp. http://c.ymcdn.com/sites/www.ser.org/resource/resmgr/custompages/publications/ser_publications/rare_species_mngmt_climate_c.pdf
138. Cowx IG, Angelopoulos N, Nunn AD (2009) Guidelines for environmentally sound practices for introductions and translocations in aquaculture. Deliverable 4.4 to the European Commission FP6 Coordination Action 044142. Environmental impacts of Alien species in aquaculture (IMPASSE) project
139. Paschos I, Perdikaris C, Gouva E, Nathanailides C (2008) Sturgeons in Greece: a review. J Appl Ichthyol 24:131–137
140. Paschos I, Nathanailides C, Kagalou I, Leka E, Tsoumani M, Perdikaris C (2003) The prospects for restoring the nearly extinct population of the Adriatic sturgeon *Acipenser naccarii* Bonaparte 1836 (Acipenseridae) in Greece. Aqua J Ichthyol Aquat Biol 7:123–132
141. Zogaris S (1999) Management of artificial water features for the avifauna. In: Hatzilacos D (ed) Concise guide: the impacts of works and activities on birds and their habitats. Hellenic Ministry of Environment, Physical Planning and Public Works, pp 238–251 (In Greek)
142. Hobbs RJ, Higgs E, Harris J (2009) Novel ecosystems: implications for conservation and restoration. Trends Ecol Evol 24:599–605. doi:10.1016/j.tree.2009.05.012
143. Moyle P (2013) Novel aquatic ecosystems: the new reality for streams in California and other Mediterranean climate regions river research and applications. River Res Appl. doi:10.1002/rra.2709
144. Crook DA, Lowe WH, Allendorf FW, Erős T, Finn DS, Gillanders BM, Hadwen WL, Harrod C, Hermoso V, Jennings S, Kilada RW (2015) Human effects on ecological connectivity in aquatic ecosystems: integrating scientific approaches to support management and mitigation. Sci Total Environ 534:52–64
145. Mimikou M, Baltas E (2013) Assessment of climate change impacts in Greece: a general overview. Am J Clim Chang 2:46–56
146. Körner C, Sarris D, Christodoulakis D (2005) Long-term increase in climatic dryness in the East-Mediterranean as evidenced for the island of Samos. Reg Environ Chang 5:27–36
147. Maas G, Macklin M (2002) The impact of recent climate change on flooding and sediment supply within a Mediterranean mountain catchment, southwestern Crete, Greece. Earth Surf Process Landf 27:1087–1105
148. Muñoz-Mas R, Papadaki C, Ntoanidis L, Martínez-Capel F, Zogaris S, Dimitriou E (2016) Generalized additive and fuzzy models in environmental flow assessment: a comparison employing the West Balkan trout. Ecol Eng 91:365–377
149. Mawdsley JR, O'malley R, Ojima DS (2009) A review of climate-change adaptation strategies for wildlife management and biodiversity conservation. Conserv Biol 23:1080–1089
150. Kondolf GM, Podolak K, Grantham TE (2013) Restoring Mediterranean-climate rivers. Hydrobiologia 719:527–545
151. Schmeller DS et al (2008) Advantages of volunteer-based biodiversity monitoring in Europe. Conserv Biol 23:307–316

Index

N. Skoulikidis et al. (eds.), *The Rivers of Greece: Evolution, Current Status and Perspectives*, Hdb Env Chem (2018) 59: 433–440,
DOI 10.1007/978-3-662-55369-5,