Society of Wetland Scientists Europe chapter

Annual Meeting *(on-line)*

16th SWS Europe Chapter conference |

June 15th – 17th 2021

Connecting wetlands functioning and biodiversity
Towards nature-based solutions

ABSTRACTS BOOK

CONTENTS

Welcome to the 16th SWS Europe Chapter and Tour du Valat Conference

The 16th conference of the European Chapter of the Society of Wetland scientists is co-organized by **Tour du Valat research Institute** and **SWS-Europe.** Initially planned to be held in Arles (Camargue, Southern France), this conference will be only online because of the still prevailing sanitary conditions and difficulties to travel.

The Tour du Valat, which is located in the heart of the Camargue, is a private research institute with the legal status of a non-profit foundation that works in the public interest. It was founded in 1954 and is dedicated to the development of scientific knowledge, the pledge for nature conservation and the sustainable management of Mediterranean wetlands. For thousands of years, the wetlands around the Mediterranean basin have provided people not only with essential services like water, food, materials and transport, but have also played a major part in their social and cultural activities. However, they have been profoundly degraded, especially during the second half of the 20th century when they were often converted to agricultural, urban or industrial lands, or replaced by artificial wetlands.

The ambition of the conference is to share results and experience from Europe and other parts of the world in order to contribute to the integration of various approaches towards the conservation of the functions and of the biodiversity of wetlands, especially in the Mediterranean region. Scientists, managers and policymakers are facing a difficult challenge, to demonstrate that the conservation and the sound use of wetlands can provide solutions for the main challenges Mediterranean societies are facing (e.g. climate change mitigation, poverty reduction, access to freshwater, nature restoration and rewilding).

The 16th SWS-Europe e-conference is meant to be a rehearsal for the 17th conference, provisionally planned for June next year, which can hopefully be organized in the classical, physical way in Arles, allowing direct communication and networking and field visits of the Camargue wetlands.

We sincerely hope that the 16th conference will be a great success. We are fortunate that we have 6 tremendously interesting key-note lectures on a variety of topics which are relevant for the objectives of the Tour du Valat and the SWS Europe chapter. In addition, we have 21 contributed papers by scientists and students. We wish you an inspiring and valuable conference!

Patrick Grillas, Matt Simpson, Jos Verhoeven

16th SWS Europe Chapter conference | on line

“Connecting wetlands functioning and biodiversity Towards nature-based solutions”

Tuesday 15th June 2021

14.00-14.15	**Introduction to the meeting (Patrick Grillas, Tour du Valat Institute)**	

14.15 - 12:30	**Session 1. Ecological studies**	
14.15 – 14.45	**Biodiversity in Mediterranean wetlands : state, threats and solutions**	**Thomas Galewski (Tour du Valat Institute, France)**
14.45 – 15.15	**Rewilding wetlands: from wetland restoration to nature-based solutions for climate change adaptation.**	**Liesbeth Bakker (Netherlands Institute of Ecology (NIOO-KNAW))**
15.15 – 15.30	Ecology and management of alpine wetlands in the climate change perspective: the CIMaE project	**Marie Lamouille-Hébert**, Florent Arthaud, Thibault Datry
15.30 – 15.45	The influence of plant functional type and phenology on photosynthesis and plant inputs to soil as affected by simultaneous changes in the environment	**Bernhard Glocker,** Jiří Mastný, Keith Edwards
15.45 – 16.00	Phylogeography in Iberian Anostraca (Crustacea: Branchiopoda) and considerations on their conservation strategies	**Lucia Sainz-Escudero**, P.C. Rodríguez-Flores, M. García-París
16.00 – 16.15	Slovenian watercourses host a variety of macrophyte communities	**Igor Zelnik**, Urška Kuhar, Matej Holcar, Mateja Germ and Alenka Gaberščik
16.15 – 16. 30	**Discussion (15min)**	
16.30 – 16.45	Degraded raised bog restoration measures and future improvement expectations based on monitoring and modelling data. An example from Madiešēni Mire in Latvia	**Līga Strazdiņa,** Māra Pakalne, Krišjānis Libauers, Rūta Abaja, Jevgenijs Fiļipovs, Dainis Jakovels
16.45 – 17.00	Redox conditions of highly saline, gypsum-rich wetland soils	**Carmen Castañeda,** Rafael Rodríguez-Ochoa, Borja Latorre, Brian Scott, Martin Rabenhorst
17.00 – 17.15	Monitoring the colouring phenomenon in the karstic lakes of Cañada del Hoyo (Cuenca, Spain) using Sentinel-2 images	Miriam Ruiz, S. Morales; J.M. Soria
17.15 – 17.30	**Discussion (15min)**	

17.30	**End of the first day $**

Wednesday 16th June 2021

14.00-14.15	**Introduction to the meeting (Jos Verhoeven, Utrecht University)** **Highlight Lake Ohrid Ramsar designation:** **Nadezda Apostolova, Tobias Salathé, Alessio Satta**	
14.15 – 14.45	**Connecting young wetland scientists with other professionals**	**Tobias Salathé (Ramsar Secretariat, Switzerland)**
14.45 – 15.15	**Rights of Wetlands: A Paradigm Shift to Meet Global Challenges**	**Gillian Davies (BSC Group, USA)**
15.15 – 15.30	Safeguarding success: Exploring community engagement strategies that support long-term restoration success in European wetlands	**Stijn den Haan**, Laszlo Pinter
15.30 – 15.45	Machine learning classification and accuracy assessment from high-resolution images of coastal wetlands	**Ricardo Martinez Prentice**, Miguel Villoslada Peciña, Kalev Sepp, Raymond D. Ward
15.45 – 16.00	Contribution of remote sensing to an overview of the heritage coastal system of a Ramsar site: case of the Kerkennah archipelago (Tunisia)	**Syrine Souissi,** Balkis Chaabene, Faiza Khebour
16.00 – 16. 15	**Discussion (15min)**	
16.15 – 16.30	The habitats of Gallocanta Lake (NE Spain) as viewed by Sentinel-1	**J. Sanz-Cano**, B. Latorre, C. Castañeda
16.30 – 16.45	Analysis of the changes in Lake Prespa and its surroundings over the last 4 decades by remote sensing methodology	**Nadezda Apostolova,** Juan M. Soria
16.45 – 17.00	Studying ponds ecosystems for climate changes mitigation: H2020 PONDERFUL project	**Maria Cuenca-Cambronero**
17.00 – 17.15	Developing DEMO-sites to highlight the efficiency of Pondscapes as Nature Based solution for adaptation and mitigation to changing climate (H2020 Ponderful project)	**Aurélie Boissezon**, Cuenca-Cambronero Maria, Bartrons Mireia, Benejam Lluis, Beklioğlu Meryem, Brucet Sandra, Davidson Thomas, Lemmens Peter, Meerhoff Mariana, Mehner Thomas, Quintana Xavier, Nicolet Pascale, Oertli Beat
17.15 – 17. 30	**Discussion (15min)**	
17.30	**End of the second day**	

Thursday 17th June 2021

14.00-14.15	**Introduction to the meeting (Matthew Simpson, President SWS-Europe, Director 35% Ltd)**	
14.15 – 14.45	**Wet grassland restoration: principles, practice and prospects'**	**Chris Joyce (University of Brighton, UK)**
14.45 – 15.15	**About smaller and bigger kidneys: riparian zones as nutrient buffers in Denmark**	**Dominik Zak (Aarhus University, Denmark)**
15.15 – 15.30	Impacts of climate Change on Wetland Plant Communities: A Mesocosm Study	**Bergamo, T**. Ward, A. D., Joyce, C.B., Sepp, K.
15.30 – 15.45	Do common reed leaves incrustations affect litter decomposition?	**Mateja Grašič**, Matevž Likar, Katarina Vogel-Mikuš, Tijana Samardžić, Alenka Gaberščik
15.45 – 16.00	Eutrophication trend evaluation in a Greek lake, using remote sensing analysis	**Irene Biliani,** Vasilis Kiousisa, Ierotheos Zachariasa
16.00 – 16. 15	**Discussion (15min)**	
16.15 – 16.30	Implementing surface flow constructed wetlands in Denmark using a new layout design	**Astrid Ledet Maagaard,** Carl Christian Hoffmann, Rasmus Jes Petersen
16.30 – 16.45	Small-scale water reservoirs as hotspots for GHG emissions	**Jelle Boode**, Jeroen de Klein
16.45 – 17.00	Fishing for methane: Ebullition is the main pathway of methane emissions from freshwater fish ponds	**Renske J.E. Vroom**, Nathan Barros, Rafael M. Almeida, Raquel Mendonça, Ive S. Muzitano, Icaro Barbosa, Ernandes S. Oliveira Junior, Alexander S. Flecker, Sarian Kosten
17.00 – 17.15	Assessment of Blue Carbon Stocks of North Bull Island, Dublin, Ireland	**Shannon Burke**, Sadhbh McCarrick, Elke Eichelmann, Grace Cott
17.15 – 17. 30	**Discussion (15min)**	
17.30	**End of the third day Matthew Simpson,**	
17.35	**Conclusions**	
17.40	**Prize to the best presentation**	
17.45	**Closure of the conference, Next meeting in Arles (June 2022)**	

Keynote 1: Biodiversity in Mediterranean wetlands: State, threats and solutions

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INTRODUCTION

The Mediterranean basin is one of the world's largest biodiversity hotspot. However, its species richness is not evenly distributed across ecosystems, with wetlands concentrating more than 30% of vertebrate diversity but covering only 2% of the terrestrial area of the hotspot (Geijzendorffer et al., 2018). The region is also a hotspot for climate change and socio-economic stresses that are exerting unprecedented pressures on the environment, particularly on wetlands, which are disappearing at a rate higher than the global average (Dixon et al. 2016). In response to this situation, civil society and governments have undertaken conservation actions for several decades.
It is essential to evaluate the conservation policies implemented in favour of wetland biodiversity in order to verify their effectiveness and propose adjustments if necessary. Such a monitoring and evaluation work is difficult due to the great divergence between countries in terms of the quantity and quality of biodiversity monitoring.
We propose here to review some of the scientific results obtained in the framework of the Mediterranean Wetlands Observatory, which contributed to the assessment of conservation policies in the region, and to the identification of possible levers to meet regional commitments to conserve biodiversity.

METHODS

To characterise the status and trends of biodiversity in the Mediterranean basin, we relied on two main datasets: the International Waterbird Census (IWC) and the IUCN Red List of threatened species. These two datasets are not specific to the Mediterranean region but specific efforts have allowed the generation and collection of a singularly large amount of data for the Mediterranean region. They are also the only source of biodiversity data for some areas, especially in the South and East of the Mediterranean. We also collected abundance time series of vertebrate species dependent on wetlands from literature. By adding these data to those of the IWC we were able to calculate a Living Planet Index for Mediterranean wetlands (Galewski et al. 2011).
In order to assess the impact of human activities and conservation policies on biodiversity, we used existing or self-created databases on several variables including land use change in selected wetlands, location of protected areas and climate variables.

RESULTS and DISCUSSION

State and trends of biodiversity in Mediterranean wetlands

Available IUCN Red List assessments of vertebrates, invertebrates and plants show that 36% of wetland-dependent species are globally threatened with extinction, a figure higher than in any other biome. Marked differences exist between taxonomic groups, which are partly the result of varying levels of endemism. The Living Planet Index shows a 28% decline for freshwater species between 1993 and 2017 with contrasted trends between taxonomic groups with birds increasing and every other groups decreasing.

Impact of anthropic activities on wetland biodiversity

The analysis of the IUCN Red List assessments shows that mismanagement of water resources, agriculture, climate change and urbanization are the main threats affecting wetland species. The indicators we have developed to measure the impact of some of these pressures reveal that agriculture is a major driving force behind the conversion of wetlands, water over-abstraction, dam construction, and water pollution (Geijzendorffer et al. 2018).

The interaction between these different drivers must be also considered. We showed that the destruction of natural wetlands, even at low levels, prevents the adjustment of waterbird communities to rising temperatures, increasing the risk of local extinction (Gaget et al. 2020a).

Evaluation of conservation policies

We found that countries have made varying efforts to achieve international biodiversity protection targets. The European Union has implemented strict regulations protecting a significant number of waterbirds as a result of its commitments under the Bern Convention on threatened species. The result is an increase in waterbird numbers in these countries (Gaget et al. 2018). The Maghreb countries have been particularly attentive to the designation of Ramsar sites covering their wetlands of international importance, which has also had a positive impact on the trend of waterbirds (Gaget et al. 2020b). On the other hand, several eastern Mediterranean countries show a deficit in the implementation of protection measures, with notably too few protected wetlands resulting in particularly worrying biodiversity trends.

Non-achievement of biodiversity conservation targets

For biodiversity, for which relatively precise international objectives exist through the Aichi targets, we observe for Mediterranean wetlands the non-achievement of these objectives. Biodiversity has continued to decline over the last 30 years and the solutions implemented have not been able to counteract this trend, except in special cases, such as waterbirds in the western Mediterranean, for which efforts were initiated a long time ago.

The pressures that directly threaten the biodiversity of Mediterranean wetlands through the degradation of their habitats are the result of indirect drivers locally well described (UNEP/MAP and Plan Bleu, 2020): human population growth, inappropriate governance, political instability, limited water resources and development choices towards economic sectors with a high impact on the environment (agro-industry, mass tourism, fossil fuels).

Regional solutions to bend the curve of wetland biodiversity loss

The protection of all the remaining wetlands must be a priority because if correctly implemented, it is an effective solution to restore biodiversity. As the majority of Mediterranean wetlands have already disappeared, it is also essential to restore a significant part of them. In addition to supporting disproportionate biodiversity, they also provide services to people that will be increasingly sought after to mitigate the impacts of climate change. These two objectives of protection and restoration fit perfectly into the concept of Nature-based Solutions. Besides, it is crucial to accelerate the ecological transition of our societies by raising awareness among consumers and decision-makers and by involving private sector in biodiversity conservation.

CONCLUSIONS

The biodiversity of Mediterranean wetlands is in a worrying conservation state. The loss and degradation of wetlands caused by the agro-industry, urban planning, tourism and industrial sectors as well as the non-respect of international commitments in favour of biodiversity explain this situation. Yet the recovery of several waterbird populations invites optimism. Protection and restoration of wetlands are part of the solutions to save wetland biodiversity and insure the resilience of Mediterranean societies.

ACKNOWLEDGEMENTS

This study is based on the collaborative work of too many people to list them all. We would like to especially thank Nadège Popoff, Anis Guelmami, Christian Perennou, Lorena Segura, Ilse Geijzendorffer, Fabien Verniest, Juliette Biquet, Eleonora Saccon, Nolan Boutry-Thivin, Frédéric Jiguet, Laura Dami and Marie Suet. We also thank the French Ministry of Ecological Transition, the Foundation TOTAL, the Foundation Prince Albert II de Monaco and the Foundation MAVA for their financial support.

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Keynote 2: Rewilding wetlands: from wetland restoration to nature-based solutions for climate change adaptation

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Intensive human use of landscapes has put strong pressures on ecosystems resulting in a global biodiversity crisis. Therefore, conservation of still existing valuable nature areas is of utmost importance, but it is not enough to bend the curve of biodiversity declines. Active restoration is urgently needed, and this certainly applies to wetlands as well. There is strong momentum for ecosystem restoration from the UN Decade of Ecosystem Restoration (2021-2030) and the EU Green Deal. The question is: how can we do this ecosystem restoration?

Rewilding is a novel ecosystem restoration approach that aims to provide more room for natural processes. It is a future-oriented restoration approach that aims to improve degraded landscapes. Natural processes include abiotic processes, such as the role of water in landscapes, for instance by relaxing water management and allowing more natural water level dynamics. Another set of natural processes is the role of animals that play keystone roles in landscape engineering, such as beavers and large herbivores.

A number of examples from The Netherlands demonstrate how the concept of rewilding can be applied in a variety of aquatic ecosystems. Human use of landscapes can constrain the application of rewilding, but even in intensely modified landscapes steps towards more natural processes are often possible.

Whereas rewilding can be applied to restore wetlands for its natural values, there are tremendous opportunities to expand the framework of restoration and look into the future how ecosystem restoration can be used as nature-based solution for climate change adaptation and mitigation. Wetlands are exemplary as ecosystems that capture carbon and sequester carbon in organic matter. Furthermore, if drainage of peat land for agricultural purposes is abandoned, and the land thus rewetted, this will strongly reduce the emission of greenhouse gasses. In the light of climate change, in large parts of Europe more irregular patterns of rainfall are predicted, resulting in flooding and drought. New wetlands can be created to provide a so-called climate buffer that can store water in times of surplus and provide water in times of drought.

By using rewilding approaches to restore and create new wetlands for the societal needs that result from climate change and combine these with an increase in natural values, a strong win-win scenario emerges that benefits society and biodiversity.

Ecology and management of alpine wetlands in the climate change perspective: the CIMaE project

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The unprecedented speed of climate change is leading to a global erosion of biodiversity. Alpine wetlands are hot spots of biodiversity and act as sentinels of such erosion due to the thermal adaptations of their constituent species. In addition to thermal and trophic stresses, the high elevation and isolation within landscapes induced slower population growth. Many species are becoming extinct or experiencing drastic shifts of their geographical distribution: contraction, enlargement or displacement.

To protect alpine biodiversity, it is essential to understand the underlying drivers of plant and animal distribution and determine what management strategies can be designed to mitigate the effects of climate change. Our project, the CIMAE project (Climatic Impact on mountain aquatic ecosystems) is based on the study of three biological groups: dragonflies, aquatic plants and frogs/tritons, which exhibit contrasted life history traits, including their capacity to disperse. They have also different strategies to cope with increased temperature and drought and decreased connectivity.

To identify the main conservation and management measures to mitigate climate change effects on these groups and preserve them on the French Alps, we need to i) improve knowledge about alpine wetland distribution by testing remote sensing tools, ii) analyze the species distributions in relation to different gradients of hydroperiod, water temperature and connectivity in 70 wetlands distributed along the French Alps and iii) model the current and future species distribution (2025, 2050, 2100). We will present the first methodological results and their implications on the CIMaE project development.

ACKNOWLEDGMENTS

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The influence of plant functional type and phenology on photosynthesis and plant inputs to soil as affected by simultaneous changes in the environment

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INTRODUCTION

Intact wetlands are providing a large range of direct (flood and drought protection) and indirect (biodiversity) high-value ecosystem services, but large amounts of global wetlands are already lost or highly degraded. The loss of ecosystem functionality due to degradation causes an estimated economic loss in the European Union of 38 billion Euros annually, increases interests and decreases human well-being. In order to adjust current management methods, plants and their response to environmental changes must be better understood.
In terrestrial ecosystems different soil related factors like skeleton size, water table, nutrient availability, gas mixture, plant and microbial diversity are closely related to each other. Changes in one factor affect all other factors and therefore, ecosystem functions and services. The question is, how different plants and different plant functional types react to single or multiple factor changes and how this could affect future wet grassland management approaches.

METHODS

Plants from two functional plant types (*Glyceria maxima* (G, competitive functional type), *Carex acuta* (C, conservative functional type)) were collected from a meadow close to Trebon, separated and transplanted to small pots to get one singular plant in each pot. The pots where then placed into basins on wooden constructions simulating three different water levels in each basin: flooded ((F) 15 cm above the soil surface), saturated ((S) water level equals soil surface) and dry ((D) water level 15 cm below the soil surface). Half of the basins were fertilized (Y) with macro and micro nutrients, the other half was not fertilized (N).
Measurement campaigns took place in July (J, ~ 16 h sunshine a day) and September (S, ~ 13 h sunshine a day). Photosynthesis was measured in vivo with a LiCOR 6400 applying a light curve up to 1000 PAR under constant CO_2 conditions and further estimated the fits (Lobo et al. 2013) on the calculated means. For determining the plant biomass allocation (BA), plants were harvested, separated into plant compartments (leaves, stem, roots, rootstock and rhizome) and dried at 60 °C for a minimum of 3 days before weighing. Parts of those dried tissues were milled and used for determining the Carbon (C), nitrogen (N) and phosphorus (P) contents by plant compartments using a CN elemental analyzer (ThermoQuest, Italy) for C and N whilst P was determined on a flow injection analyzer (FIA, Lachat Instruments, USA). Root exudates were collected by a modified culture based method (Edwards et al. 2018),

separated by ion exchange chromatography (IEC; Thermo ICS-5000, USA) using a dionex capillary column and a conductivity detector for organic acids and other anions and a Pac-10 column with amperiometrically detection for sugars and amino acids. The particular quality and quantity of each compound was investigated by comparing retention time peaks of known standards and peak integration.

RESULTS and DISCUSSION

Photosynthesis

In our experiment, nutrient availability was the limiting factor for photosynthesis with *Glyceria* achieving slightly higher values than *Carex* while the water level did not have a significant influence on the photosynthesis performance. Under nutrient enriched conditions the water level seems to be a limiting factor for both *Carex* and *Glyceria* whereas *Carex* achieved higher photosynthesis values under dry conditions and *Glyceria* under flooded conditions. The displayed photosynthesis rates were similar in both July and September (so only July is displayed, see Fig. 1 and 2 for *C. acuta* and *G. maxima*, respectively). Therefore, if the

Management Aim is to increase photosynthesis per leaf area in nutrient rich environments, *Glyceria* is to be preferred in generally flooded meadows, but *Carex* is to be preferred in generally drier conditions. In nutrient poor conditions no notable difference in photosynthesis was observed.

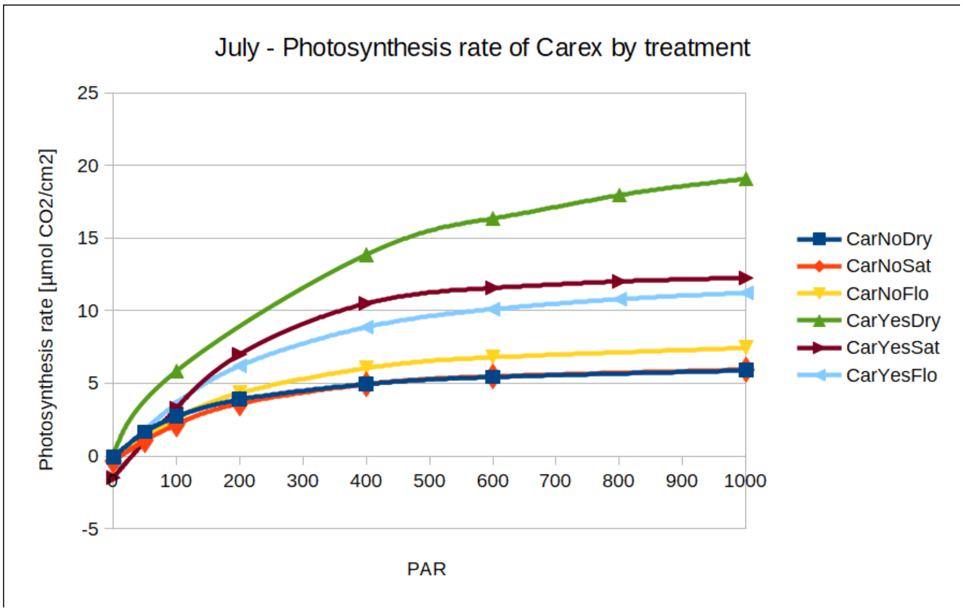

Figure 1: Estimated photosynthesis rates in Carex (Car) under different light conditions (PAR) by treatment: No/Yes additional fertilization under dry (Dry), saturated (Sat) or flooded (Flo) conditions.

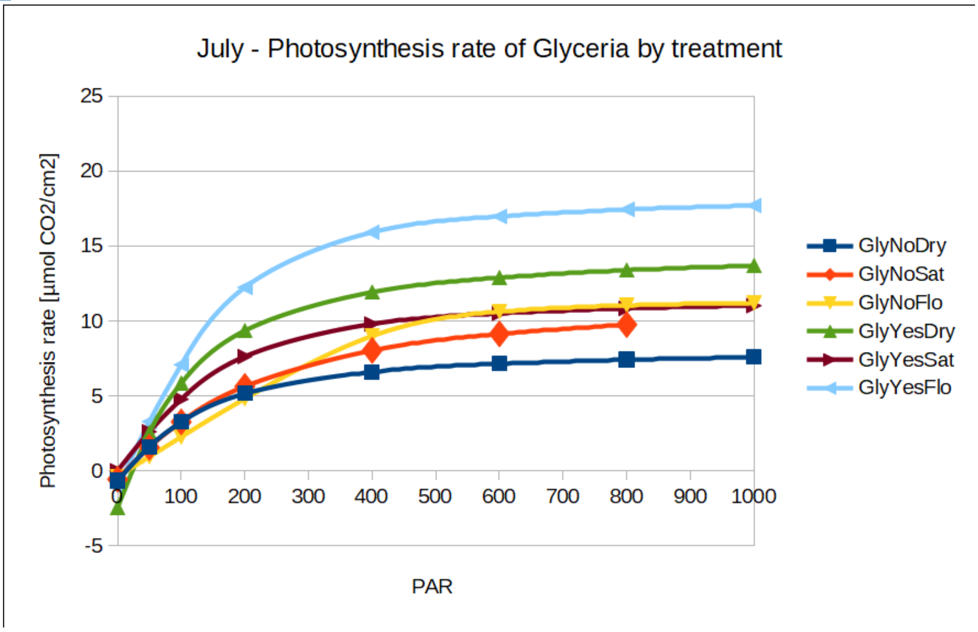

Figure 2: Estimated photosynthesis rates in Glyceria (Gly) under different light conditions (PAR) by treatment: No/Yes additional fertilization under dry (Dry), saturated (Sat) or flooded (Flo) conditions.

Root exudates

The root exudate (Table 1) measurements found that the amount of the root exudates per root area is either higher in Glyceria or not species dependent. Two exudates (Quinic acid and Propionic acid) are higher under nutrient poor conditions but for all other compounds nutrient availability showed no significant effect. Water level affected the amount of Acetic acid (highest under saturated conditions) and Oxalic and Citric acid (both higher in dry conditions, weakly significant). The total amount of most of the exudates changed over time with generally more exudates in July (longer days in July lead to more photosynthesis per day and therefore, to a higher possible investment in root exudates). However, the amounts of Succinic acid and Iso-citric acid increased in September. Succinic acid can be associated with root development (Yosikawa et al. 1993) and increased Iso-citric acid implicates increased energy metabolism (Popova et al. 1998) drawing the conclusion that the plants are already preparing for winter and re-sprout in spring. If the **Management Aim** is to have plants with more energy stored in the roots for re-sprouting in Spring, cutting should be avoided in September until leaf senescence.

Biomass Allocation

Biomass allocation assessments showed that *C. acuta* produced more leaf biomass and total above ground biomass than *G. maxima*, but *G. maxima* produced more below ground biomass resulting in higher root:shoot ratios in *G. maxima*. However, it has to be noted, that *G. maxima* hardly reached its usual heights, but replaced most of the pot volume with root stock and rhizomes. Therefore, one reason for the unexpected low above ground biomass in *G. maxima* could be too small pots.

Table 1. Overview of the measured root exudates and their dependency to various factors. Underlined increasement shows weak significance. Empty cells showed no significant difference.

Compound	Species (S)	Fertilization (Fer)	Water (W)	Month
Quinic		0,04 (N>Y)		<0,001 (J>S)
Acetic	<0,001 (G>C)		*0,087 (S>D>F)*	0,021 (J>S)
Propionic	0,001 (G>C)	*0,056 (N>Y)*		<0,001 (J>S)
Formic	<0,001 (G>C)			<0,001 (J>S)
Isobutyric	0,002 (G>C)			
Butyric	0,042 (G>C)			
Pyruvic	0,011 (G>C)			
Adipic	*0,059 (G>C)*			<0,001 (J>S)
Succinic				0,003 (S>J)
Oxalic			*0,09 (D>F>S)*	
Citric			*0,07 (D>F>S)*	<0,001 (J>S)
Iso-citric				<0,001 (S>J)

CONCLUSIONS

Although *G. maxima* is a competitive plant species and faster growing (especially shown for the below ground biomass) than the conservative *C. acuta*, there are vast meadows mostly consisting *C. acuta*. Therefore, there must be additional effects denying *G. maxima* to invade *C. acuta* dominated meadows and replace *C. acuta* as the dominant species. One reason could be higher grazing pressure on *G. maxima*, or the higher stand density in Carex. However, this is just speculation and needs further investigation. A more general comment on wet grassland management is, that in nutrient rich conditions *G. maxima* is more productive in flooded environments, but *C acuta* is more productive in drier areas of a wet grassland. Regardless the environmental conditions, *G. maxima* is releasing more root exudates which potentially affect the soil microbia. For increased plant fitness in Spring, cutting should be avoided in September until leave senescence so both plant species can properly restore their energy deposits in the roots.

ACKNOWLEDGEMENTS

Hereby, we want to thank the Grant Agency of the Czech Republic and the University of South Bohemia for financial support and providing the location to run the experiments. Moreover we want to thank our amazing colleges and friends, who supported us all the way along from setting up and maintaining the experimental site to harvesting the plants making life much easier in those difficult times.

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Phylogeography in Iberian Anostraca (Crustacea: Branchiopoda) and considerations on strategies for their conservation

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INTRODUCTION

Species diversity and their distribution ranges are two valuable elements in determining the biological richness of a specific area or system, and essential to recognize priority areas for conservation. However, the genetic diversity component is frequently neglected. The species' genetic diversity is the allelic or genotypic richness present in the populations connected by gene flow that conform a species. Phylogeography is the tool that allows visualizing how this diversity is distributed across space by using genetic and GIS data. The different nature between mitochondrial and nuclear genomes, and the variable mutation rates of their genes, enable to recover information about species such as their evolutionary history, and also historical and current demographic patterns. These data are useful in terms of evolutionary knowledge, but also in conservation management, identifying the evolutionary units and evaluating their preservation status. In this work, we apply phylogeographic approaches in the study of the genetic diversity of the two Iberian species of Anostraca (Crustacea: Branchiopoda) *Branchipus schaefferi* and *Branchipus cortesi.* Anostracans are crustaceans that inhabit continental temporal aquatic habitats such as wetlands lakes but also small ponds which form in semi/impermeable substrates. We analyse the allelic diversity of two genetic markers in order to obtain their distribution pattern in a spatial framework and to test gene flow among populations. These data allow us to understand the dispersal strategies of these species, to improve our knowledge about their evolutionary history and to recognize ancestral lineages and intraspecific conservation units.

METHODS

We use genetic data of the mitochondrial and nuclear markers *coxI* and *ITS2*, respectively, to perform phylogeographic analyses. We construct haplotypes networks with sequences of the two species by using informatics programs that allow to align our dataset, obtain the collapsed matrix of unique alleles and create the networks.

RESULTS and DISCUSSION

The evolutionary history of these two species has been clarified through the mitochondrial genetic marker, unveiling the ancestral lineages. The nuclear marker has allowed to obtain the gene flow among populations. The geographical distribution of the allele diversity of the two anostracan species have been obtained through the phylogeographic haplotype networks.

From these results we can infer the dispersal dynamic followed by these populations and the existence of possible isolated ones (little or absence of gene-flow with other populations).

CONCLUSIONS

Phylogeographic studies are useful to discover the intraspecific diversity of a species and manage adequate conservation actions. Anostracans construct metapopulation systems that enable them to keep their populations connected by geneflow. These metapopulations are conformed by large and natural aquatic habitats in wetlands and intermediate ponds that keep populations connected. These minor habitats are less considered in conservation but play a key role in the preservation of Anostraca species and populations.

Slovenian watercourses host a variety of macrophyte communities

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INTRODUCTION

Rivers and streams are heterogenous ecosystems that host a great number of vascular plant communities. The territory of Slovenia is highly diverse regarding geomorphologic, geologic, climatic, and edaphic conditions. Slovenia is also situated on the junction of four biogeographic regions, namely the Alpine, Dinaric, Pannonian and Mediterranean regions. High environmental diversity of Slovenia results in high biodiversity. We presumed that environmental variability will also affect the distribution of hygrophilous vascular plants in running waters and consequently the structure of plant communities they form.

In this study, we investigated 34 Slovenian watercourses belonging to hydro-ecoregions the Po lowland, Dinarides, and the Pannonian lowland, in order to examine growth forms of macrophyte species and macrophyte communities in these watercourses, and to point out the potential regionality of these communities. We also expected that environmental variability would affect number of taxa, their abundance, and the occurrence of different growth forms.

METHODS

We analysed macrophytes, spatial, and environmental parameters in 906 stretches of the watercourses occurring in the Dinaric, Pannonian, and Po lowland hydro-ecoregions. Macrophyte species abundance was estimated as a relative plant biomass using a five-degree scale, namely 1 – very rare, 2 – rare, 3 – commonly present, 4 – frequent, and 5 – predominant (Kohler and Janauer 1995). These values were transformed by the function x^3, as suggested by Schneider and Melzer (2003). We differentiated the following growth forms; pleustophytes (pl), submerged anchored species (sa), floating-leaved species (fl), amphiphytes (am), and helophytes (he) (Janauer et al. 2018).

The assessment of environmental conditions was performed according to modified Riparian, Channel, and Environmental (RCE) Inventory, (Petersen 1992) that comprised 11 parameters. We also assessed current velocity according to Janauer et al. (2018) using a 4-level scale (1 – no visible current, 2 – slow, >0 to 30 cm s^{-1}, 3 – medium, 35 to 65 cm s^{-1}, and 4 – fast current, >70 cm s^{-1}).

RESULTS and DISCUSSION

We determined 87 vascular plant taxa. The most abundant were *Myriophyllum spicatum*, *Phalaris arundinacea*, and *Potamogeton nodosus*. Submerged macrophytes presented about one third of total species abundance, while amphiphytes were somewhat less abundant. Canonical Correspondence Analysis (CCA) revealed that distance from the source explained 15.1% of the growth form type variability, and current velocity and latitude explained 4.1% each. With the assessed parameters, we explained 31.6% of the variability. When CCA was run with taxa, only 20.9% of their variability was explained with statistically significant parameters. We distinguished 25 different plant associations belonging to five classes and nine alliances. The majority of defined plant communities were distributed in different watercourses belonging to different hydro-ecoregions. Only seven communities had a narrower distribution range, three of them on karst poljes. Among them, the new association *Mentho aquaticae-Oenanthetum fistulosae* from the river Mali Obrh on the Loško polje was described in this contribution.

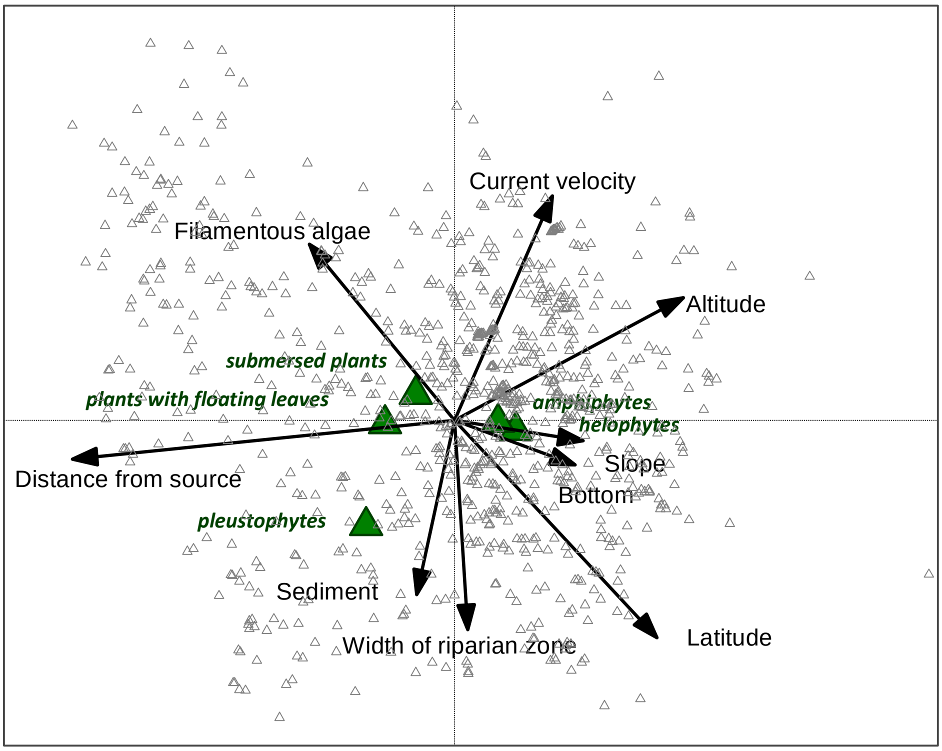

Fig. 1. CCA plot showing the relationships between the abundance of different growth forms and significant environmental parameters detected in different stretches of the examined rivers (from Zelnik et al. 2021, Water, 13, 8, 1071).

Correlation analyses revealed positive relationships between species diversity and retention structures, sediment and detritus deposition, while there were negative relationships between filamentous algae abundance and current velocity and slope. In addition, negative relationship was also obtained between species diversity and distance from the source, while positive relationship was found between species diversity and altitude (Table 1).

The majority of the defined communities are distributed in various watercourses belonging to different hydro-ecoregions. However, seven out of 25 communities had a narrower distribution range. Among these were the association *Myriophylletum verticillati*, found only in the river Struga (NE Slovenia), the associations *Potamo crispi-Ranunculetum trichophylli* and *Schoenoplectetum lacustris*, which were mostly limited to watercourses in karst poljes, the association *Polygonetum hydropiperis* that occurred only in 7 stretches in the lower part of the river Ščavnica (NE Slovenia), the association *Eleocharito palustris-Hippuridetum vulgaris* that was one of the rarest and occurred only in the river Ižica in central Slovenia, the association *Glycerio notatae-Veronicetum beccabungae*, which was recorded only in 4 stretches of the upper part of the river Temenica in central Slovenia, and the newly described association *Mentho aquaticae-Oenanthetum fistulosae* that was recorded only in 5 stretches of the river Mali Obrh on Loško polje, which is also a karstic polje

Table 1. Spearman correlation coefficients between diversity, abundance, and assessed environmental variables. S-W – Shannon-Wiener diversity index; *– very highly significant (p <0.001), ** – highly significant (p <0.01)**

Variable	No. of taxa	S-W	Abundance
Distance from the source	-0.314***	-0.304***	-0.215***
Altitude	0.383***	0.363***	0.175***
Slope	-0.141***	-0.132***	-0.167***
Land use	0.084**	0.097**	0.115***
Width of riparian zone	ns	ns	ns
Completeness of riparian zone	0.123***	0.131***	0.133***
Vegetation of riparian zone	0.072*	0.078*	0.131***
Retention structures	0.188***	0.169***	0.104**
Sediment deposition	0.196***	0.206***	0.184***
Riverbed bottom	0.207***	0.180***	0.202***
Flow dynamics	0.120***	0.129***	0.121***
Presence of detritus	0.216***	0.189***	0.192***
Current velocity	-0.365***	-0.322***	-0.386***
Presence of filamentous algae	-0.205***	-0.191***	ns

CONCLUSIONS

High heterogeneity of the surveyed watercourses, their catchments, as well as environmental parameters resulted in high diversity of vascular plants and consequently in high diversity of plant communities they form. We found high heterogeneity of aquatic vegetation and low total share of species with a wide ecological valence. We determined a total of 87 vascular plants among which 36% of total abundance belonged to submerged macrophytes, 30% to amphiphytes, 18% to helophytes, 3% to floating-leaved plants and only 3% to pleustophytes. The most abundant species was *M. spicatum* with 12% share, followed by *P. arundinacea* and *P. nodosus*. Spatial parameters explained the highest share of species presence and abundance variability. The examined river stretches host 25 different associations belonging to 5 classes and 9 alliances, which are mostly distributed in different hydro-ecoregions. Seven plant associations had local distribution, and three of them were only found on karst poljes. This distribution pattern reveals specific environmental conditions in these karst poljes and lack of regionality elsewhere.

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Degraded raised bog restoration measures and future improvement expectations based on monitoring and modelling data. An example from Madiešēni Mire in Latvia

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INTRODUCTION

Augstroze Nature Reserve (total area 4007.2 ha) includes four raised bogs and other types of wetlands (lakes, transition mires and quaking bogs, bog woodlands, swamp forests), representing 16 different habitats of European Union importance (Ikauniece (ed.), 2019). Large part of the Nature Reserve is taken by Madiešēni Mire (area 1881 ha), which is an excellent example of active raised bog (7110*), and it is among the largest raised bogs in Latvia.

The Northern part of Madiešēnu Mire was drained in a time period from 1983 to 1998. Drainage has caused increased decomposition and compaction of peat, especially near ditches, and increased establishment, growth and productivity of trees. Degraded raised bog vegetation has overtaken the drained area.

Water level and water level fluctuations play a major role in mire ecosystems. One of the simplest and more effective methods to implement habitat restoration and hydrological regime optimization in raised bogs is to reduce the drain-induced negative impacts on the local ecosystem of the mire. Taken the appropriate corrective actions, which have been focused on the negative impact mitigation, it is possible to stop or significantly reduce the degradation of ecosystems. One of the key actions to achieve this goal is full or partial backfilling of existing drainage ditches. However, too high-water levels may reduce plant productivity, which negatively impacts on peat formation (Joosten, 1993).

Based on hydrogeological modelling results, to restore the degraded area in Madiešēni Mire around ditches in total area of 147.8 ha, 26 peat dams were built in October 2020 to prevent further degradation of the peatland ecosystem and its functions.

Habitat and site hydrology monitoring is necessary to show the effect of management actions. The monitoring has been started prior to the practical management implemented and will be carried out after it. To survey changes of vegetation composition induced by drainage, additional research was conducted using remote sensing data. Vegetation classification based on high resolution hyperspectral data were used to prepare scenarios of habitat restoration success and relate vegetation to greenhouse gas (GHG) emission volumes.

METHODS

Study area

Augstroze Nature Reserve comprises four raised bogs (Madiešēni Mire, Lielkalni Mire, Vecmuiža Mire and Bisnieki Mire), one transition mire (Namītēni Mire) and three lakes (Augstrozes Lielezers Lake, Dauguļu Mazezers Lake and Bisnieki Lake), fairly large forest areas, and some patches of grasslands and other habitat types. The site includes both intact peatlands (active raised bog habitats (7110*)), as well as areas damaged by drainage (in the northern part) where degraded raised bog habitats still capable of natural regeneration (7120), as well as various forest types occur. The active raised bogs are characteristic with hummock-hollow complex and labyrinth of bog pools.

Thickness of peat in Madiešēni Mire is averagely 4.0 meters. Peat is predominantly composed of poorly decomposed *Sphagnum fuscum* peat. The most common plant species in the tree layer are *Pinus sylvestris* and *Betula pubescens*, in the dwarf shrub layer – *Ledum palustre*, *Calluna vulgaris*, *Rubus chamaemorus*, *Oxycoccus palustris*, *Andromeda polyfolia*, in the herb layer – *Drosera rotundifolia*, *Eriophorum vaginatum*, *Rhynchospora alba*, *Trichophorum cespitosum*, in the bryophyte layer – *Dicranum bonjeanii*, *Mylia anomala*, *Pleurozium schreberi*, *Sphagnum angustifolium*, *S. fuscum*, *S. magellanicum*, *S. rubellum*, and the lichens *Cladina stygia* and *Cladonia* spp.

Hydrological monitoring

To evaluate the efficiency of restoration, hydrological monitoring using wells (piezometers) is implemented. Information from a piezometer can be used to determine vertical flows of water i.e. whether the peat is recharging an underlying aquifer or if water is being discharged from the aquifer to the peat. The depths of the wells (pipes) allows measuring the groundwater table throughout the year. Generally, natural groundwater fluctuations in peatlands and adjacent areas do not exceed 1–1.5 m, whereas near drainage ditches the range of fluctuations may be larger than 2 metres. To cover this range, the piezometers installed in Madiešēni Mire reach a depth of 2.5 m (PVC pipes with length of 2.5 m and 40 mm diameter). From this length c. 0.5 m is above ground. The lower part of the pipe (1 m length) is perforated, both ends of the pipe are closed with PVC covers.

The piezometers are located at different distances from the ditches where building of dams was performed (transects perpendicular to the ditches). In Madiešēni Mire, two transects were established, each of them with three piezometers which are located perpendicular to drainage ditches where dam building is planned. Another three piezometers are located in relatively intact raised bog – two of them close to ditches, one – in the intact mire.

Each piezometer is equipped with an automatic data logger, so called pressure logger. They measure the groundwater table each hour, i.e. 24 times per day. The data are automatically stored and are manually downloaded from each piezometer once per 3–4 months. Storing of data in each logger can last for about 24 months.

Vegetation monitoring

Vegetation is monitored in square-shaped sample plots of size 100 m^2 (10 x 10 m). In each plot, parameters like relief, soil properties and pH, land use history, tree, shrub, herb, bryophyte cover, perimeter of trees are characterised. For detailed description of vegetation, the large plot is subdivided into small units, 1 x 1 m internal subplots, in total nine within each large square. Vegetation is described by using Braun-Blanquet method: all vascular plant, bryophyte and lichen species and their cover is recorded in percent (visual estimation).

Vegetation monitoring sample plots were located close to the transects of groundwater monitoring wells (at least, near one of the groundwater monitoring transects) and greenhouse gas measurement sites. In Madiešēni Mire, four sample plots were established, out of these 2–3 plots in the degraded part close to drainage ditches and one in a relatively intact part of the mire.

Remote sensing technology

Remote sensing data of the project area was collected on 03.06.2018. with an airborne surveillance and environmental monitoring system (ARSENAL) that includes aircraft equipped with hyperspectral data sensors, LIDAR and high resolution visual camera. From the hyperspectral data, 27 most informative spectral bands of visible and near infrared spectral region were obtained for vegetation analysis. A LIDAR point cloud was used to estimate vegetation structure differences and visual high resolution images were acquired for validation purposes. During summer, two peatland habitat and restoration specialists collected field data in the project area of vegetation classes as reference and data for further remote sensing data analysis. Final process was to prepare maps and, using indirect GHG emission estimation methodology (Couwenberg et al., 2011; Koska et al., 2001), to model different scenarios of restoration effect to vegetation composition, CO_2, CH_4 emission volume and global warming potential (GWP).

RESULTS and DISCUSSION

Groundwater table

Visual estimation of increasing groundwater table level in ditches and wetter areas around the dams five months after management actions indicates tendencies of re-establishment of stabilized hydrological conditions in degraded part of Madiešēni Mire. It was affirmed by direct groundwater table measurements which showed that greatest changes after dam building are in wells No. 1, 2, 4 and 6 which are located closest to the restoration area (Fig. 1). Not only the relative groundwater level but also more moderate level fluctuations, if comparing the situation before and after management, indicate a successful restoration process.

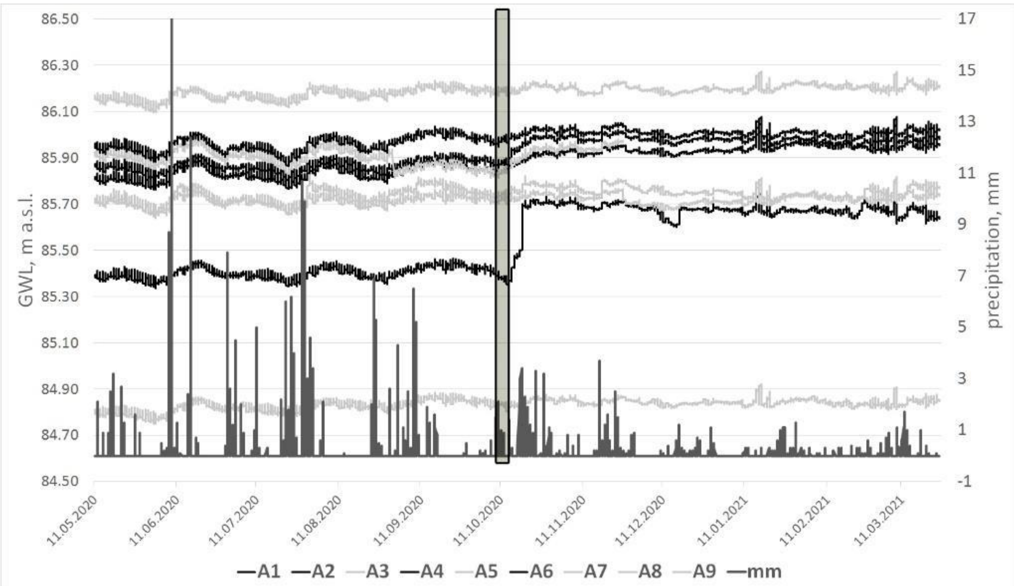

Fig. 1. Groundwater table level (GWL) five months before and after restoration in Madiešēni Mire in Augstroze Nature Reserve. Measurements from wells with significant changes or increasing

tendencies of GWL are in black color. Time of dam building is marked as a grey rectangle in the middle of diagram.

Modelling of vegetation response to management using remote sensing technologies

Direct observations of vegetation composition after dam building using monitoring plots has been impossible yet due to the winter-spring season. However, after visiting the site five months after management actions confirmed the expectations. Vegetation cover has fully closed around the building area as the re-established moisture level in peat surface has positive effect on vascular plant and bryophyte growth. More detailed analysis is planned in the summer season to evaluate spreading of *Sphagnum* species in the more degraded area. Vegetation in the project area was mapped using hyperspectral data sensors and LIDAR, and summarized as species groups. For most of these groups a quantitative estimation of GHG emissions was applied using the GEST methodology. In the future, as suggested by post-restoration development in other raised bogs, the cover of *Sphagnum* spp., *Oxycoccus palustris* and other raised bog species will increase, whereas *Calluna vulgaris*, other dwarf shrubs as well as bryophytes of dry coniferous forests (like *Dicranum bonjeanii, D. polysetum, Pleurozium schreberi*) will decrease as the stabilized hydrology improve occurrence of moisture-loving plants. If the groundwater level rise will be rapid after building of dams, withering of trees and *Calluna vulgaris* is expected.

It was assumed that tree cover in the research site in the 1st post-restoration scenario will remain in current condition, whereas the ground vegetation will recover to a condition similar to the intact raised bog. In case that groundwater level rise is rapid and the trees cannot adapt to the new conditions, tree die-off is possible, as predicted in the 2nd post-restoration scenario. Calculated GHG emissions in either case scenario within the next 50 years are by one half or even smaller than in baseline situation without restoration measures.

Before the management, the number of species per plot and subplots was rather low – on average 20.0 species per plot and 9.8 species per subplot. High increase of species richness is not expected as the raised bog environment is suitable for species adapted to this wet habitat.

CONCLUSIONS

Active management of degraded raised bog habitat has a momentary positive effect on groundwater level. Improvement of vegetation composition and mitigation of GHG emissions which is directly related to plant species occurrence will follow over the longer term.

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Redox conditions of highly saline, gypsum-rich wetland soils

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INTRODUCTION

Wetland soils with aquic conditions and hydric soils have been identified using redoximorphic features based on the release of iron and manganese in saturated and reducing soil conditions. The observation of redoximorphic features associated with soil wetness is the most common method to infer the reduction of soils. Those indicators are key in many international soil classification systems and are used for jurisdictional delimitation of wetland boundaries in the USA for protection purposes.

However, the identification of reducing conditions based on field indicators and morphologic features is still difficult for some types of soils due to soil texture and/or specific soil composition. To overcome the difficulties associated with determining the reducing character of soil, the indicators of reduction in soils (IRIS tubes), which are coated with iron and manganese oxides, are used. IRIS technology, developed by Jenkinson and Franzmeier (2006) and adapted by Rabenhorst and Burch (2006), has been accepted as a standard indicator of soil reduction by the NTCS committee of hydric soils in the USA (Rabenhorst, 2012). The principle of IRIS technique interpretation is that removal of the iron or manganese paint coating shows areas of reduction, and the more coating that is removed, the greater the degree of reduction within the soil.

In arid regions, soils accumulate carbonates, gypsum and soluble salts. Identifying aquic conditions and hydric soils is difficult because redoximorphic features are not well expressed due to the high pH, low amounts of iron and manganese, and low microbial activity. These wet saline soils are present throughout arid and semiarid Mediterranean regions, some of them located along flyways crucial for the survival of migratory bird populations. Others are home to endangered bird populations or endemic plants.

In the Central Ebro Valley (NE Spain), the water saturated soils of Monegros saline wetlands are subject to the evapoconcentration of salts caused by saline groundwater and the dissolution of salt-bearing geologic materials. The predominant limestone and gypsum-rich bedrocks condition soils that are naturally low in iron and high in pH, and the presence of a salt crust is yet another clear indicator of soil saturation. Few data are currently available on how saline and alkaline soils affect iron reduction and anaerobic conditions. This work aims to explore the IRIS technique to identify reducing conditions in highly saline gypsum- and carbonate-rich soils of the Monegros saline wetlands. This study will increase our knowledge of wetland soils and improve our ability to manage and protect them.

METHODS

Study area

Salineta playa lake (41º 28.9' N, 0º 9.6' W), with a surface area of ~23 ha, is part of the endorheic complex of the Monegros saline wetlands (locally named saladas) in NE Spain. Salineta is excavated on carbonate-rich lutites, at 325 m a.s.l., and register the highest flooding frequency and water salinity of all the Monegros saladas (Castañeda and Herrero, 2005), with salt contents in excess of 400 g L^{-1}. The water occurrence of this playa-lake varied from 80% to 98%, as obtained from weekly field data (1993-1997) and from 52 Landsat images (19842004), respectively.

The extent of vegetation is limited to the salada fringe and includes five habitats listed in Annex I of the 1992 EU Directive (92/43/EEC): 1310 Pioneer annual vegetation with *Salicornia* and other species from muddy or sandy areas, 1410 Mediterranean saline grasslands (*Juncetalia maritimi*), 1420 Mediterranean and thermo-Atlantic halophilic scrubs (*Sarcocornietea fruticosae*), 1430 halonitrophilic scrubs (*Pegano-Salsoletea*), and 1510 Mediterranean salt steppes (*Limonietalia*) (Conesa et al., 2010).

IRIS setting and auxiliary data

Two plots (1 and 2) about 30 m away and with a difference in elevation of ~0.06 m were selected in the playa area of the wetland. Four sets of five replicate IRIS tubes each, totaling 20 iron and 20 manganese tubes per plot, were inserted into the soil using a push probe in August 2017. Each plot included a piezometer and two pairs of ceramic suction probes installed at 20 and 40 cm depths.

All the IRIS tubes except for one manganese set were retrieved on nine dates (one set of five tubes per sampling), from 05/09/2017 to 21/12/2017. The last manganese set was retrieved in May 2018. The tubes were systematically photographed in the field under vertical view. A single digital image of the whole surface of the tube was obtained and digitally processed. We quantified the area with depleted iron or manganese oxide coating from the IRIS tubes (Rabenhorst and Burch, 2006; Castañeda et al., 2017). For each set of IRIS tubes, we calculated the averaged percent area of paint removal in the total tube length (50 cm).

Water level measurements were taken on the dates the IRIS tubes were retrieved, and rainfall data were collected from the nearby Bujaraloz weather station. Field and laboratory physical and chemical determinations of soil and water characteristics were carried out. Electrical conductivity (EC), pH and redox potential were measured with a field probe. Gypsum content was quantified by thermogravimetric analysis and calcium carbonate equivalent (CCE) was determined by the Bernard calcimeter method. Major ions were determined by gas chromatography (Metrohm 861), dissolved iron and manganese were determined by atomic absorption.

RESULTS and DISCUSSION

Physical and chemical soil and water characteristics

Soils are Gypic Aquisalid with a gypsum content of 54% and 13% of CCE. The mean soil temperature from August to November was 16º C (st. dev. 6.0). The soil was neutral, with a mean pH = 7.0, and EC of 196.3 dSm^{-1} at 25º C (st. dev. 69.4). The mean Eh was 112 mV, with more than 56% of measurements indicating reducing conditions. Following the criteria in Barlett and James (1985), the soil presented predominant sulfidic conditions with some suboxic periods.

The soil solution composition was of chloride-sulfate sodic magnesian type, with a mean electrical conductivity of 163.2 dSm^{-1} at 25º C and pH = 7.2. Water type and ionic composition were consistent with data obtained in previous works, showing that the saline wetland has preserved its hypersaline character in recent decades. The piezometric water was of similar composition with slightly less calcium content (Table 1). On average, piezometric levels were about 30 cm (±15) depth.

The contents of dissolved iron and manganese (Table 2) were very low and similar in soil water and piezometric water. This could be related to the fluctuating Eh and pH values that prevent the persistence in the water of the soluble forms of these ions. Moreover, the carbonate content in the soil and the sulfate-reducing conditions restricted the presence of iron- and manganese-soluble ions.

Table 1. Mean (standard deviation) of the pH, electrical conductivity and ionic content (meq L^{-1}) of soil and piezometric water.

Solution	pH	EC dS m^{-1}	Ca^{2+}	Mg^{2+}	Na^{+}	K^{+}	$NH4^{+}$	$SO4^{2-}$	Cl^{-}	HCO^{3-}	$NO3^{-}$	Br^{-}
Soil water	32.5 (12.4)	163.15 (7.0)	27.1 (9.8)	1751.9 (242.2)	1895.1 (217.1)	71.8 (7.3)	ip	584.4 (285.0)	3122.1 (342.5)	4.0 (0)	ip	1.2 (0.1)
Piezometric water	7.1 (0.2)	162.1 (8.8)	21.9 (2.3)	1857.8 (237.0)	1918.1 (257.1)	73.9 (8.1)	ip	630.5 (293.6)	3218.5 (371.1)	4 (0)	ip	1.3 (0.2)

Table 2. Mean (standard deviation) of the iron and manganese content (meq L^{-1}) of soil and piezometric water.

Solution	Fe^{2+}	Mn^{2+}
Soil water	0.66 (0.14)	0.63 (0.39)
Piezometric water	0.62 (0.04)	0.57 (0.20)

Iron and manganese oxide removal from the IRIS tubes

Table 3 shows the results of plot 2 because a decay in activity was observed in plot 1, especially for iron oxide, despite their proximity. It evidences the spatial variability of soil geochemical conditions, probably due to the complex hydrology. As expected, manganese oxide mobilized faster than iron. The maximum removal of manganese oxide was 88% for the 50-cm tube and was reached after two months and 48 mm of rain. At that time, the iron oxide was only 5.5% depleted in the tubes. The four subsequent extractions of iron oxide tubes showed a low rate of depletion, and after 138 days (May 2018) the iron oxide depletion was only 40%.

The removal rate for the manganese tubes was very constant (Table 3) at about 9.7% per week, in spite of the low rains registered in the area during the study period. The total removal of iron oxide was low during autumn and winter, less than 9%. This low rate (<1% per week) slightly accelerated in spring (2% per week), probably associated with the increase in spring temperature and soil water content. Rainfall significantly increased on the last sampling date, with 84 mm of accumulated rains prior to the last sampling. The highest rate of iron oxide depletion in Salineta is comparable to that obtained in carbonate rich soils of Gallocanta Lake, about 1.7% per week (Castañeda et al., 2017). In both wetland soils, besides the removal of iron and manganese oxdes, black mottles on the IRIS tubes were formed indicating the precipitation of iron sulfides. The percentage of depletion with depth (Fig. 1) showed that maximum reducing activity took place at the subsurface soil layers, deeper for iron than for manganese.

Table 3. Sample dates of the iron and manganese IRIS tubes, 30-day previous accumulated rainfall (P30), percentage of oxides depleted in the 50-cm tube (%Mn, %Fe), number of days (D) and weeks (W) elapsed, and removal rate (% per week).

Sample date, Mn	P30 (mm)	%Mn 50 cm	D	W	Mn rate (%)	Sample date, Fe	P30 (mm)	%Fe 50 cm	D	W	Fe rate (%)
12/09/2017	15.8	51.6	38	5.4	9.5	05/10/2017	10.6	5.5	61	8.7	0.63
14/09/2017	22.0	59.0	40	5.7	10.4	06/11/2017	24.2	9.8	72	10.3	0.94
20/09/2017	20.0	62.4	46	6.6	9.5	21/12/2017	2.6	8.5	93	13.3	0.64
05/10/2017	10.6	88.1	61	8.7	10.1	03/05/2018	84.6	40.2	138	19.7	2.04

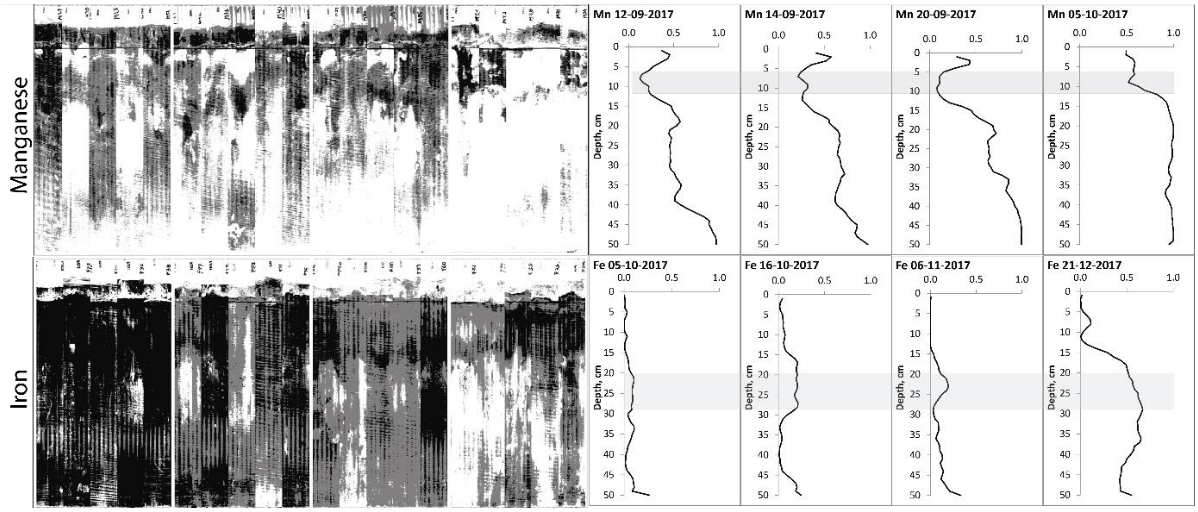

Fig. 1. Composition of binary images of depleted IRIS tubes and corresponding percent of paint removal with depth. Gray bands indicate the depth with maximum depletion.

CONCLUSIONS

The quantification of iron and manganese removal using the IRIS method has been evidenced for the first time in hypersaline, gypsum-rich wetland soils under intermittently flooding conditions. Low rates of iron and manganese oxide removal were obtained due to the soil composition and the non-permanent water flooding character associated with aridity. The results were key to designing new experiments aimed at determining the factors that condition the soil ecology and biogeochemistry of arid wetlands.

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Monitoring the colouring phenomenon in the karstic lakes of Cañada del Hoyo (Cuenca, Spain) using Sentinel-2 images

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INTRODUCTION

Inland surface waters form valuable ecosystems (Buchanan and Stubblebine, 1962) of great importance for global biodiversity (Wunder, 2015). Karst lakes are lentic ecosystems, with waters rich in bicarbonates (Camacho *et al*., 2009), which develop on sedimentary rocks. As they are linked with areas where karstification phenomena are very active (Camacho *et al*., 2009), these types of lakes frequently have rounded shapes, steep sides, lack of freshwater inflow or outflow and a relatively large depth. Therefore, the waters are often thermally stratified, giving rise to water compartments with different physicochemical and biological conditions. (Casamayor *et al*., 2012). Due to the oligotrophic characteristics of the waters, they allow the development of communities dominated by charophytes. Underwater meadows of *Chara* spp. are a good bio-indicator of water quality as their populations decline when water nutrient richness increases. (Cirujano *et al*., 2007).

The Cruz Lake, presents a biogenic meromixis, due to the enrichment of the monimolimnion water (the deep layer not mixed which remains without any kind of exchange with the upper waters, except for slow diffusion processes (Izhitskiy *et al*., 2016)) with calcium, magnesium, and iron bicarbonates (Miracle *et al*., 1992). These physicochemical changes are evident every summer, at the end of July, when the blue-green waters of the lake acquire a milky appearance (figure 1) due to calcium carbonate precipitation (CMADR, 2007) favoured by high temperatures, high pH and/or an increase in primary production. (Vanderploeg *et al*., 1987; Homa and Chapra, 2011). In Cañada del Hoyo there are other lakes that also show impressive colour changes like Lagunillo de las Tortugas, which acquires a reddish colour (figure 1) and Cardenillas in greenish color.

The aim of the work is to carry out a monitoring of the whitish, reddish and greenish colour phenomena of the karstic lakes of Cañada del Hoyo through a thematic analysis of the optical properties of the water bodies using Sentinel-2 satellite images. In this way, it is possible to collect updated data on the colouring phenomena of the small lakes, e.g. the starting dates and duration in time.

METHODS

The present study is developed on the site known as “Lagunas de Cañada del Hoyo”, catalogued as a natural monument in 2007. They are located southeast of Los Palancares, in the karst area called Los Oteros, in the central-eastern part of the province of Cuenca (Spain). It consists of a group of dolines, seven of which maintain water permanently and are known as: Cruz, Tejo, Lagunillo, Parra, Llana, Tortugas and Cardenillas. (CMADR, 2007).

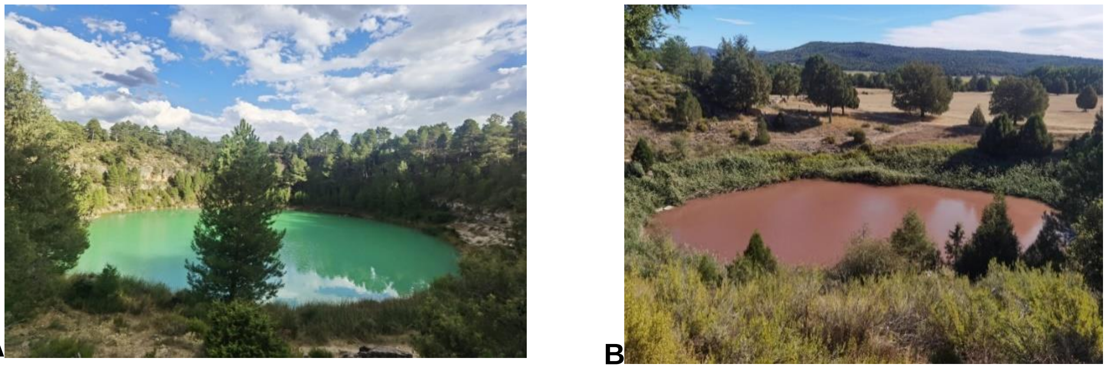
A B

Fig. 1. Comparison of the two colours of the lakes. (A) picture of the La Cruz during the whiting phenomenon in summer of 2020. (B) Tortugas Lake during the reddish colour in autumn 2020.

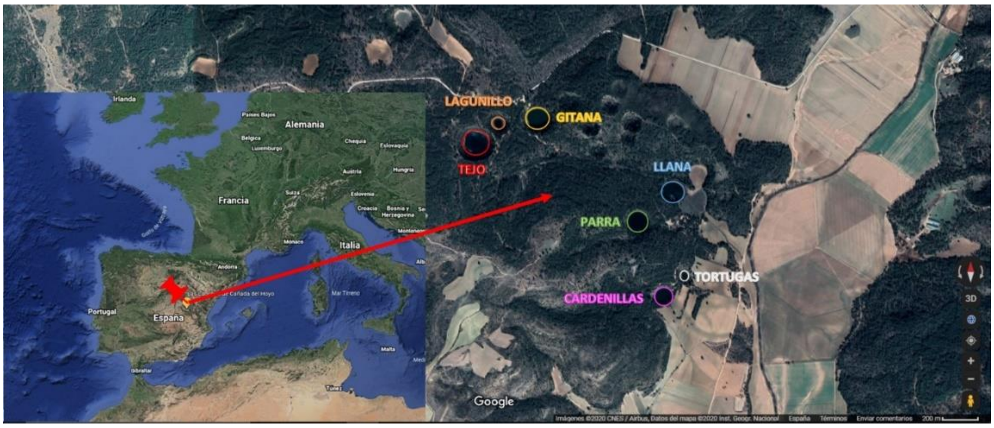

Fig. 2. Location of the Cañada del Hoyo karstic complex in the Iberian Peninsula. The seven lakes are highlighted in a circle with their respective names. Next to them there are other dolines with no water at the bottom (Source: modified from Google Earth).

Monitoring the phenomenon:

The monitoring of the "colouring phenomenon", as the exclusive optical property of the three lake, was carried out through the use of satellite images from the Sentinel-2 platform of the European Space Agency, obtained from the Copernicus Open Access Hub repository and subsequently processed with the SNAP satellite image processing software.

The Copernicus download platform holds images from the start of the Sentinel-2 operation in 2015 to the present day. The images were downloaded with level 1 processing (L1C). Atmospheric correction was performed by means of the C2RCC SNAP module using C2X as the methodology. The images were resampled and cropped (10 m pixel resolution) using the geographic coordinates as the boundary area 40.00 N; -1.90 W; 39.97 N; -1.85 W. Bands 2, 3 and 4 were selected to display in false colour (RGB) the colouring phenomenon.
At the same time, data on atmospheric temperature and rainfall were collected at the Cuenca weather station during the dates covered by the study period.

RESULTS and DISCUSSION

A total of 59 Sentinel-2 images have been downloaded through the Copernicus Open Access Hub. Meteorological data have also been obtained for the months of July and August in order to study the relationships between recorded temperatures and rainfall and the whiting of the water. The appearance of the whiting is sudden, it is preceded by a period of high temperatures that take place from eight to fourteen days before. The months of July and August are the periods of maximum temperatures of the year in this region. Hence, these are the months where the whiting phenomenon happens yearly. The decay of the white phenomenon is slower and takes a few days longer, as it is a process of sedimentation of particulate matter.
In the five years studied, the white phenomenon has not appeared more than once per year. Nor is it mentioned in the work carried out since 1980 on La Cruz lake that there have ever been two periods. This should indicate that the whitening of the lagoon is on an annual cycle.
A remarkable fact is the length of time the lagoon remains white. According to the data collected from 2015 to 2019, the phenomenon lasts at least approximately one week (Miracle *et al*., 1992; Rodrigo *et al*., 1993) and the maximum would be 18 days, considering that Sentinel-2 images have been collected every three to five days. It is important to note that the lack of satellite images available for some years makes it difficult to count the exact days when the lagoon presents the phenomenon as in 2015.
The reddish phenomenon is related to anoxia in the Tortugas lake (figure 3). The coloration appears as a consequence of the mixing of the lagoon in autumn and the presence of anoxic water at the bottom that reaches the surface. The red-coloured sulphur bacteria remain as long as the water remains without oxygen. This period lasts for several months, the last of which began in September 2020 and continues six months later. In other years it has not occurred, or its duration has been shorter, as in 2018 which lasted the months of December to February.
The greenish phenomenon is due to phytoplankton growth in Cardenillas lake. The coloration appears in late summer when microscopic algal growth occurs in the water body. The green colour of the water is not as spectacular as bleaching or reddening. However, it is striking to the visitor that two lagoons a few meters apart, one is greenish and the other reddish. This coloration does not last beyond the month of November, when algal growth stops due to cold temperatures and the waters regain transparency.

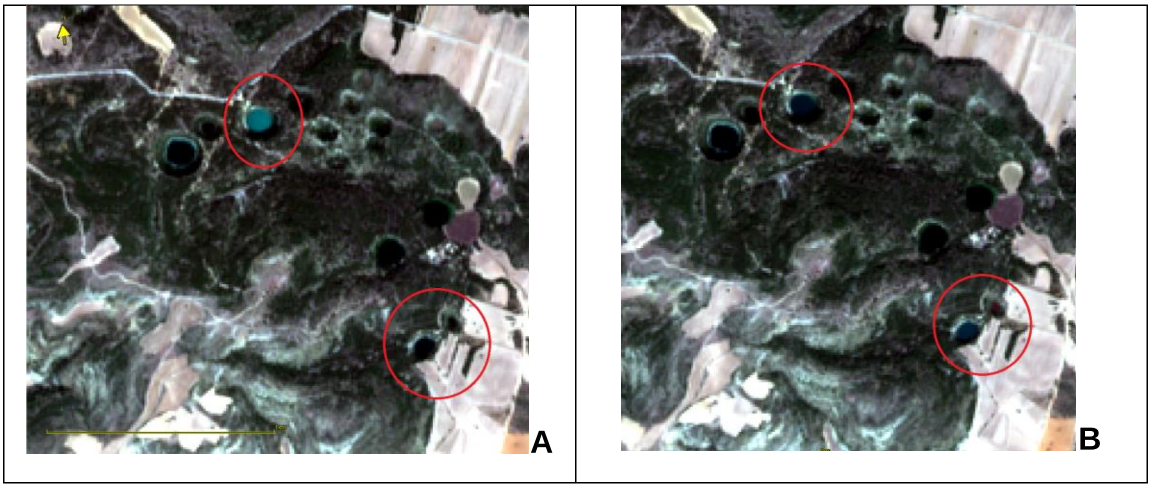

Fig. 3. Sentinel-2 image in RGB false colour. (A) August 26, 2020 of La Cruz lakes during the whiting phenomenon. (B) September 12, 2020, Cardenillas and Tortugas lakes during the greenish and reddish colour.

CONCLUSION

The use of the SNAP tool as the processing software for Sentinel 2 satellite images and the processing of satellite images resulted in a relatively simple and efficient desktop work, since, in spite of presenting a multitude of different applications, its interface is intuitive and hardly generated any problems or doubts during its use. The availability of Copernicus' images is an essential factor for research work of this type, which combines remote sensing and limnology. The results obtained show the usefulness of the application to a practical case of basic science somewhere between geochemistry and biology, which is very difficult to investigate under field conditions. It is striking how three lakes so close to each other in the same system show such spectacular colour changes in a short period of time.

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Keynote 3: Connecting young wetland scientists with other professionals

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The SWS provides a forum for wetland scientists to interact with other professionals for the exchange and mutual benefit of respective knowledge, know-how and data. This requires professionals who are willing to learn to know and understand each other, and to get informed about both the underlying science and the considerations that affect decision-making, based on research results. Young scientists are particularly concerned, because it is they who need to develop proposals how to solve the problems they are inheriting from the ruling generation. An important first step to transfer research results is cooperation between different scientists. It often brings together related disciplines. However, it should go beyond related disciplines, and link different fields of natural, economic, historic and social sciences. Such dialogue and cooperation between different professionals is only still at its beginning. Nevertheless, wetland issues require this dialogue and cooperation, because wetland ecosystems provide the nexus that links the different fields that are important for sustainable development. Finding lasting solutions to wetland problems needs crosscutting research cooperation. Dialogue requires a common language among participants from different fields. A common language creates understanding by all professionals, including laypersons. To reach mutual understanding, the inter-professional dialogue should start with a focus on basic issues and principles. This helps to create a mutual agreement on basic values shared by the actors from different fields.

The current pandemic illustrates the challenge that scientists face when trying to be understood and trusted by non-scientists and the wider public. Creating understanding and adherence to scientific concepts, ideas and proposals by other professionals and laypersons is difficult, but necessary. Researchers trying to achieve this, should distinguish between two complementary parts of information they need to explain: first presenting the facts and concepts that are widely agreed within the scientific community, then distinguishing those facts from issues where research has not yet been able to remove remaining uncertainties. Understanding wetland ecosystem services belongs to the category of well-established facts; at least for wetland scientists. They are studying these interactions since more than fifty years, when the term 'wetlands' was created to address all water-related terrestrial and coastal-marine ecosystems. However, this concept is not yet well known nor understood by other professionals. Equally, wetland scientists argue that wetland-based solutions provide multiple benefits for different users. Again, this is yet still little known nor agreed by other professionals.

These are sufficient reasons to encourage wetland researchers to engage more in dialogue between science and policy and to increase their efforts to communicate their findings and recommendations to decision-makers. To this end, researchers can work with allies such as the European Union that promotes and develops instruments for both fields: science and policy. However, professionals in either field hardly meet nor work together. The process how research findings are transformed into policies is obscure, complex, and often guided by sectoral, economic interests. This unsatisfying situation provides opportunities for young researchers to bridge the important gaps and to get involved. Wetland scientists can also use the intergovernmental treaty on wetlands that countries agreed upon during a conference at the Caspian seaside resort of Ramsar fifty years ago, on 2 February 1971. The 'Ramsar Convention' provides a framework for the implementation of nature-based solutions. It develops policy instruments and implementation tools for practitioners. Their quality depends on the quality and diversity of the inputs that wetland scientists are contributing and willing to explain to national delegates who develop tools for wetland conservation and restoration. SWS researchers are encouraged to engage actively with the experts of the Ramsar Scientific and Technical Review Panel at national, regional and international levels.

Keynote 4: Rights of Wetlands: A Paradigm Shift to Meet Global Challenges

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Human-caused climate destabilization and biodiversity loss pose existential threats to wetlands, ecosystems and society, and healthy wetlands are essential to reversing these global challenges (Finlayson et al 2018, Moomaw et al 2018). Time is short (Bradshaw 2021). The next few decades will determine our collective fate (Ripple et al 2017, Ripple et al 2020, Trisos 2020). Efforts to reverse degradation and loss of wetlands have failed (Davidson 2014, Davidson et al 2020), despite the efforts of the Ramsar Convention on Wetlands, established in 1971, and many national and sub-national efforts. These failures call for new approaches, including a fundamental ethical and legal paradigm shift that recognizes the legal and living personhood of wetlands (Davies et al 2020).

Indigenous peoples, local communities and non-profits have been leading a growing global Rights of Nature movement that reconsiders the human-Nature relationship and returns to a recognition of Nature's living beingness and inherent rights, a perspective that has been shared by many cultures and societies throughout history, and particularly by many Indigenous cultures and local communities (Nash 1989, Cullinen 2011, Koons 2012, Wilson 2019). This shift in perspectives (Kimmerer 2013) moves the human-Nature relationship from one of exploitation, depletion, degradation and loss to one based on relational values such as reciprocity, gratitude, and respect. Such a paradigm shift may lead to greater success in achieving conservation, restoration, re-wilding and remembering our integrated and relational presence as a part of Nature.

Through the SWS Ramsar Section and Rights of Wetlands Initiative, a group of wetland and climate scientists and attorneys proposes a *Universal Declaration of the Rights of Wetlands* (Davies et al 2020, Simpson et al 2020, Davidson et al 2021, Fennessy et al 2021). This *Declaration* is being shared globally (see https://www.rightsofwetlands.org/) with Ramsar Convention Signatory Countries, non-profits, and others, and thus far has been endorsed by 25 organizations, including SWS, Wetlands International, Wildfowl and Wetlands Trust, Society for Ecological Restoration, and the Community Environmental Legal Defense Fund. The SWS Rights of Wetlands Initiative is seeking support from Ramsar Convention Signatory Countries for a Ramsar Draft Resolution on Rights of Wetlands.

This presentation will articulate the context for the proposed *Declaration*, outline what the *Declaration* entails, identify how it differs from existing rights of Nature declarations, and address how the *Declaration* can be utilized to further conservation, protection and restoration of wetlands globally.

ACKNOWLEDGEMENTS

I would like to acknowledge that the land that I am writing from in Acton, Massachusetts, USA is the unceded tribal territory of Nipmuc and Nashoba Tribal People, and I wish to thank the elders and ancestors for permitting my presence on these traditional lands.

Nipmuc means People of the Fresh Water and Nashoba means Wolf. I would like to acknowledge the leadership of many Indigenous Peoples and local communities in sustaining recognition of the living beingness of Mother Earth and elements of Nature, and the importance of basing the human-Nature relationship on one of gratitude, reciprocity, and respect.

This presentation is based on the work of my co-authors and members of the SWS Ramsar Section and Rights of Wetlands Initiative who have contributed significant, time, energy, thought and creativity in support of Rights of Wetlands, particularly Nick Davidson, Siobhan Fennessy, Max Finlayson, Royal Gardner, Rob McInnis, Bill Moomaw, Erin Okuno, Dave Pritchard, Matt Simpson and Jack Whitacre. I would like to thank them for their efforts and good cheer during this past pandemic year, during which the *Declaration* was forged.

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Safeguarding success: Exploring community engagement strategies that support long-term restoration success in European wetlands

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The effective engagement of local communities is essential to safeguard the long-term success of ecosystem restoration efforts. However, research efforts on understanding the effectiveness of community engagement strategies has predominantly focused on the community-based restoration of tropical forests, which has left other ecosystems understudied, including freshwater wetlands which also harbor high levels of biodiversity and act as carbon sinks.

By interviewing project managers, we explored community engagement strategies employed by nine previously completed freshwater wetland restoration projects across Europe, which were funded by the EU LIFE program between 2011 and 2015.

Qualitative content analysis of these interviews revealed the use of consultative and co-productive typologies of community engagement strategies. Furthermore, the perspective of the project managers emphasized five most important community engagement strategies to support the long term success of restoration projects. These included (i) partnering with local governmental bodies to build upon pre-existing networks of trust; (ii) effectively empowering local communities and integrating their knowledge in decision-making processes; (iii) using context-adapted, relevant education of key stakeholders on the benefits of restoring biodiversity; (iv) building effective personal relationships through one-on-one meetings; and (v) maintaining high levels of engagement by starting similar restoration projects in adjacent areas.

Possible explanations for the effectiveness of these community engagement strategies in safeguarding restoration success are discussed using the Theory of Participation (Reed et al. 2018). These community engagement strategies may guide practitioners to enhance the long-term effectiveness of future wetland restoration efforts in Europe and beyond, for the benefit of the planet and its people.

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[NB: this is a work-in-progress and conclusions may slightly change]

Machine learning classification and accuracy assessment from high-resolution images of coastal wetlands

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INTRODUCTION

Plant communities have been used as indicators to study the structure of semi natural coastal wetlands present around the Baltic Sea (Burnside et al., 2007; Berg et al., 2012). For large areas of coastal wetland these studies can be difficult because of the heterogeneous plant composition and the inaccessibility to some areas.
In the last decade, Unmanned Aerial Vehicles (UAV) have provided an excellent tool for remote sensing of ecology and biodiversity. They provide multispectral and high-resolution images of relatively large areas to undertake rapid environmental assessments (Villoslada Peciña et al., 2021). These images provide the necessary data to perform a classification of the habitat using reflectance values for each pixel of the images.
Several studies have applied Machine Learning (ML) algorithms to obtain the highest accuracies in supervised classifications of high-resolution images although there is no agreement on which performs better because it depends on the study area (Maxwell et al., 2018). The most common task is to classify the pixels of the image but recently, more and more studies are applying another approach for classification which consists on classifying groups of homogeneous pixels called objects which hold spectral and spatial information of the bands (Blaschke, 2010). This method has been used in many research areas, especially in wetlands (Dronova, 2015).
In spite of the increased use of ML algorithms in both pixel and object classification approaches, there is a lack of studies that comparing these classifications using high-resolution images from UAVs.
In this study, we performed a classification of high-resolution images based on two ML algorithms. The main objectives were to: (1) classify plant communities in coastal wetlands in the Baltic Sea area using two different ML algorithms from high-resolution UAV images (2) compare the classification accuracies of pixels and objects (3) perform a quality assessment between each classification map.

METHODS

Plant communities and study sites

Using the nomenclature of the phytosociological categories of plant communities (Burnside et al., 2007), we plotted three to five plant communities using a stratified random sampling in six different study areas. The plots were geolocated using the Sokkia GSR2700 ISX dGPS according to the methodology used in Ward et al. (2013).

Image acquisition and processing

High-resolution images were collected using the Sequoia multispectral camera on board a senseFly eBee fixed wing UAV. Four spectral bands (Green, Red, Red Edge and Near Infrarred) with reflectance values were processed and combined in each image to extract ten different Vegetation Indices (VIs).

Classification and accuracy assessment

The VIs and the Digital Elevation Model (DEM) were used for each study site to classify pixels and objects using two different machine-learning algorithms: Random Forest (RF) and K-Nearest Neighbour (KNN). The former has been successfully used in coastal wetlands to classify the pixels in different study areas (Villoslada et al., 2020) and the latter has been shown to be a robust classifier widely used for vegetation mapping (Chirici et al., 2016). The confusion matrices from each classification gave an overall accuracy in addition to a Kappa statistic. To compare the classifications, the percentage of area agreement and disagreement in addition to two indicators, allocation (difference in allocation of labeled categories in the study area) and quantity (difference in proportion of labeled categories in the study area) of change in maps, were applied (Jr & Millones, 2011). The remaining proportion corresponds to the chance agreement. Comparisons were performed between RF classification of pixels and objects, RF and KNN classification of objects and RF and KNN classification of pixels.

RESULTS and DISCUSSION

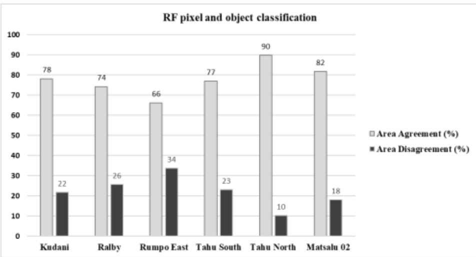
The overall kappa of all classifications was greater than 0.7 and the overall accuracy was greater than 79%, meaning a good level of classification. In all study areas, the RF algorithm performed better than KNN, both for pixel and object classifications (Table 1). There was not a clear difference between pixel-based and object-based classifications of RF although the pixel-based classification appeared more suitable because it better represents the heterogeneity in the distribution of plant communities for each study area. The proportion of the allocation and quantity disagreement between the RF and KNN classifications of pixels and objects were higher than the classification of pixels and objects with RF (Figure 1) although there are differences among the study areas, depending on the variation of the composition in plant communities due to different activities.

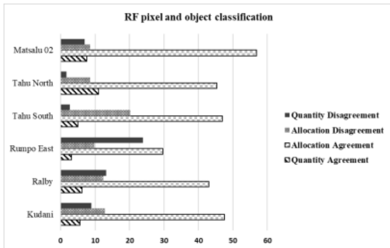
The proportion of disagreement areas was lower for the classification of pixels and objects with RF and higher for the rest of classifications (Figure 1). These areas match ecotones where different species are mixed together and thus the reflectance and their corresponding values of VIs are classified differently. This might be because of two reasons: first, the different assignation of class labels for individual pixels by the classifiers and second, because the objects homogenize the values so the classifiers interpret them as different plant communities.

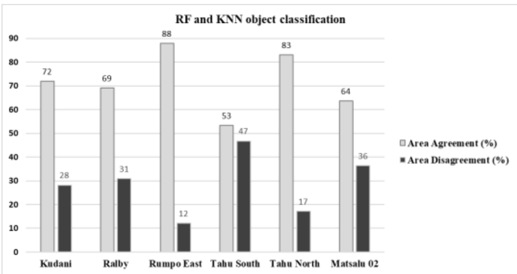
Table 1. Results of each classification for pixels and objects using Random Forest and KNN.

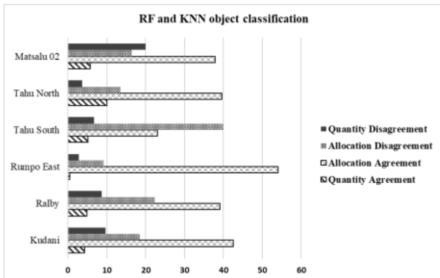
	Pixel-based classification				Object-based classification			
	Random Forest		KNN		Random Forest		KNN	
Study Area	kappa-Index	Accuracy (%)	kappa-Index	Accuracy (%)	kappa-Index	Accuracy (%)	kappa-Index	Accuracy (%)
Kudani	0.9	93	0.81	87	0.92	94	0.87	90
Matsalu-02	0.94	95	0.84	80	0.98	98	0.74	80
Ralby	0.88	91	0.91	93	0.86	89	0.8	85
Rumpo East	0.99	99	0.96	94	0.99	98	0.86	91
Tahu North	0.99	99	0.96	97	0.97	98	0.92	95
Tahu South	0.94	96	0.79	85	0.86	90	0.72	79

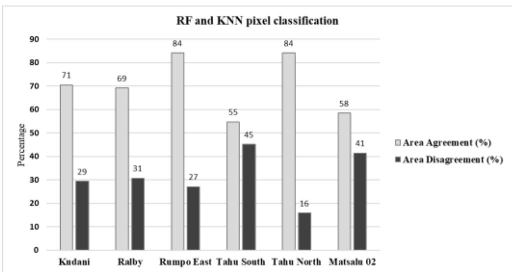
Figure 1. Comparison between classifications. a-c: Proportion of disagreement and agreement areas in each study area; d-f: proportion of disagreement and agreement of categories found in each study area. The remaining percentage corresponds to chance agreements.

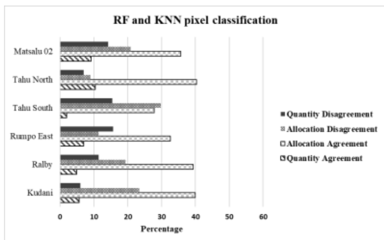












CONCLUSIONS

According to our results, we can conclude that the use of two robust machine learning algorithms to classify pixels and objects from high-resolution images:

1. Enable classification of plant communities with a high accuracy.
2. Provide better results with the classification of pixels but in all cases, Random Forest classifies better than KNN.
3. Retrieve lower scores of disagreement between areas and categories comparing the use of Random Forest for the classification of pixels and objects although this also depends on the structure of plant communities in each study area.

ACKNOWLEDGEMENTS

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Contribution of remote sensing to an overview of the heritage coastal system of a Ramsar site: case of the Kerkennah archipelago (Tunisia)

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INTRODUCTION

In 1996, IUCN initiated a project that focuses on wetlands and marine sites including island systems in the world heritage list to offer a broader view of protection and management as "*wetlands and marine areas are among the most productive natural environments on earth".* Kerkennah archipelago in the southern Mediterranean shore, is a Ramsar site (2012) characterized with low topography and depressions (Sebkha) which occupies 37% of its territory (Rhouma et al, 2005). Most of the studies have looked at the Sebkha characteristics separately as a threat to the archipelago (Etienne et al. 2012; Fehri, 2011), but did not investigate the potential of combining the adjacent goods. Much of the wetland is associated with cultural heritage sites: the *Charfiya* (intangible world heritage of UNESCO, 2020) and the palm grove (intangible world heritage of UNESCO, 2019) which are being degraded due to the extension of the Sebkha and salinization, major environmental problems of the archipelago (L. Etienne et al. 2015). To prevent further loss and implement policies for preservation, a large-scale monitoring and integration of adjacent goods is necessary. Incomplete dataset curtails the ability of experts to investigate both cultural and wetland heritage in order to build a resilient ecosystem. to address this problem, we need additional sources of sites data. Remote sensing has been an efficient adopted method for land cover mapping and monitoring and has been used in many recent wetlands research areas (Slagter et al., 2020; Ludwig et al., 2019; Guo et al., 2017)
Here, we present a study on the use of the sentinel-2 data for coastal system mapping. We first designed a classification using NDVI, IB and NDWI indices considering two land cover types: Sebkha and palm grove than we digitized the *charfiya* from a google earth image and we combined the results in one map to discuss the integrated valorization of these three goods.

METHODS

The methodological flow is discussed in details in following subsections. First, we collected data (satellite images), then data processing was done.

1. Data

A Sentinel-2 image recorded in 01-03-2020 have been studied in order to delimit the areas occupied by the sebkhas and the palm grove in the Kerkennah archipelago (available via

Copernicus; https://scihub.copernicus.eu/). The *charfiya* was digitalized from a Google earth Image.

2. Data processing

The treatments were carried out from spectral bands 4, 8 and 11 with 10m resolution. The Qgis 3.16 software was used to calculate three indices: NDVI, IB and NDWI (Table 1) and thanks to a good knowledge of the field, we were able to define the different classes of land cover obtained. Among the lands observed, emphasis was placed on the village of Mellita to locate the *Charfiya*. The results thus obtained, have been crossed in order to highlight the connection between the Sebkhas, the palm grove and the Charfiya.

Table 1. Overview of Sentinel-2 indices used for this study

Index	Description	equation
NDVI	Normalized Difference Vegetation Index: provides information on the density of plant cover and the quantity of green biomass using NIR (band 8) and RED (band 4) channels (Hountondji et al. 2004)	NDVI = (NIR – RED)/(NIR+RED)
NDWI	Normalized Difference Water Index: used for remote sensing of vegetation liquid water that uses NIR (band 8) and SWIR (band 11) channels (Gao, 1996)	NDWI = (NIR SWIR)/(NIR+SWIR)
IB	Brightness Index: Sensitive to the soil glow related to its humidity and the presence of salt. It uses RED (band 4) and NIR (band 8) channels	IB = $\sqrt{(RED^2+NIR^2)}$

RESULTS and DISCUSSION

The results obtained from the calculation and the superposition of the indices show two main classes of land use classes: the sebkhas with their holomorphic soil components, halophilic plants and hydromorphic soils, a variability of the humidity gradient is also registered, and the palm grove. The conformity of this classification was verified by comparison with the study carried out by Étienne et al, 2012 using Landsat TM 5 images. The connection between the different Sebkha types, which represent wetlands, and the palm grove is highlighted in figure 1.

Figure 2 highlights the coastal system of the Kerkennah archipelago as a support for a frequency of goods with cultural and natural heritage values. This natural richness of the region constitutes the important elements that define the landscape of the archipelago and provide substantial socio-economic benefits to the local population, especially through traditional fishing which is the economic pillar of the archipelago of which the *charfiya* are the tool and the palm grove is the raw material, as well as biotope for numerous species such as nesting birds and the fields of *Posidonia oceanica*. While the Palm grove and the *Charfiya* can often be used to address this challenge, it is enhanced when integrated with the Sebkha as it is the engine of a well hydrofunction on the archipelago.

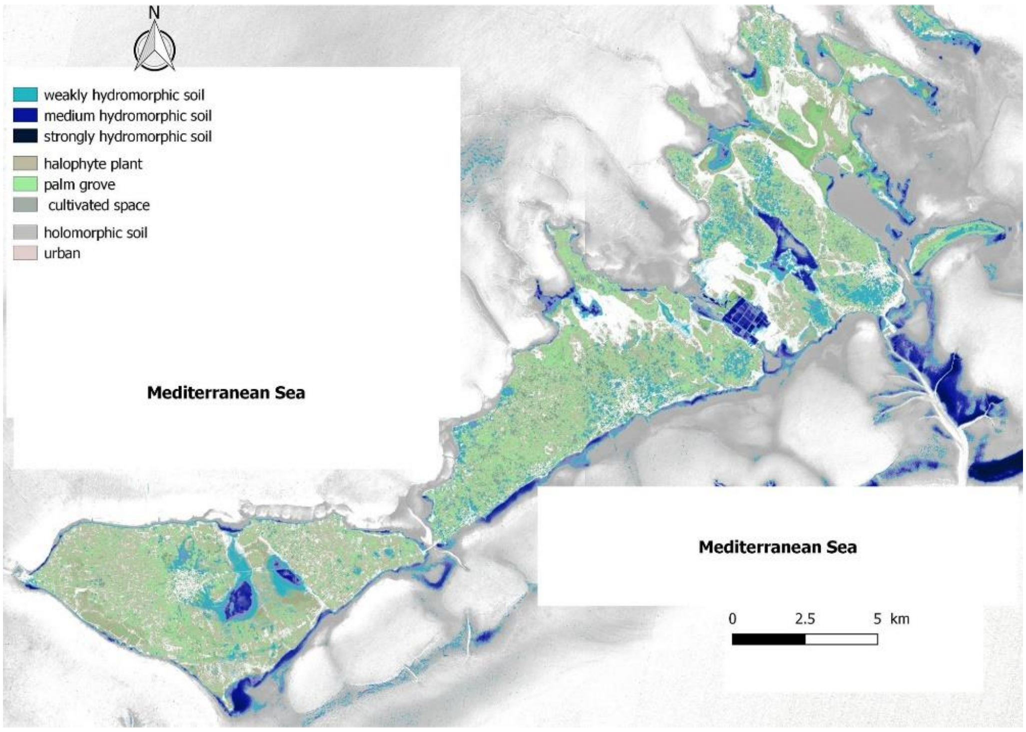

Fig. 1. Land cover classification on the Kerkennah archipelago

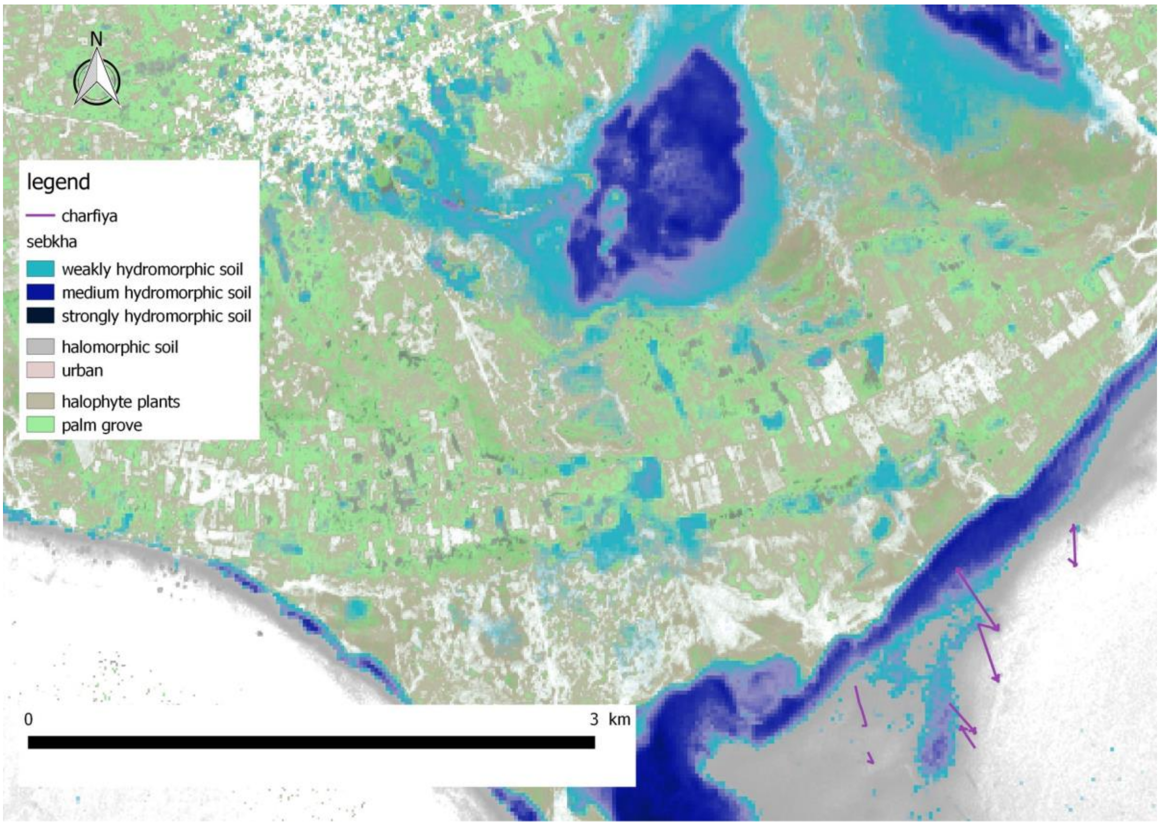

Fig. 2. Overview of the coastal system in the village of Mellita

CONCLUSIONS

The palm grove, the Sebkha and the *charfiya* constitute a heritage landscape sequence. The integrated valorization of this coastal system helps building a resilient ecosystem, since these goods are structurally and functionally interconnected, the degradation of one contributes to the degradation of the other and vice versa. The contribution of remote sensing made it possible to visualize the coastal heritage system of the Kerkennah archipelago. However, despite its usefulness, remote sensing remains insufficient and the quality and value of this system depends mainly on the perception of the local population. For this reason, social surveys remain essential for a more reliable study.

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The habitats of Gallocanta Lake (NE Spain) as viewed by Sentinel-1

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INTRODUCTION

Gallocanta Lake (40º50'N, 2º11'W) is an intermittent saline wetland located at 1000 m a.s.l. in a semi-arid environment in central Spain. It is the largest saline lake in Western Europe (14 km^2) and one of the most important ecosystems on the Iberian Peninsula. It is included in the List of Wetlands of International Importance of the Ramsar Convention, and declared a Site of Community Importance and Special Area of Conservation in the European ecological network (Natura 2000 Network).

The natural reserve encompasses 4550 ha, 44.5% of which corresponds to intermittently flooded terrestrial habitats (Castañeda et al., 2020). The habitat distribution around the lake reflects the soil moisture and salinity gradient, current geomorphological processes, and lake water level fluctuations. Water persistence is crucial for maintaining the bird populations and to supporting them along their migratory routes, so monitoring this variable is essential.

Obtaining a continuous record of water level in the field is difficult due to the complexity of accurately determining the deepest topographic point and any changes in the microtopography of the bed caused by water and wind action. Remote sensing, an alternative method for regularly monitoring flooded areas, has frequently been applied to wetlands, particularly synthetic aperture radar (SAR) systems since these are independent of weather conditions and solar illumination (White et al., 2015). However, there are few SAR studies applied to monitoring semi-arid wetlands and their habitats.

The objective of this study is to explore Sentinel-1 radar data to analyse the backscatter response of the Gallocanta Lake habitats. Remote monitoring of intermittently flooded areas at adequate temporal and spatial scales will be of great value in terms of lake management and preservation, and will allow the identification of water surface changes.

METHODS

We selected two Sentinel-1 images representative of contrasting conditions at the lake, i.e., non-flood (16/10/19) and flood (25/02/20), representing the dry and wet seasons, respectively. The images were downloaded from https://scihub.copernicus.eu/ with a pixel size of 10 m in two polarisation modes, VV and VH. The radar scenes were cropped, radiometrically calibrated, and transformed into backscatter coefficients (sigma nought, σ^0) in dB. We applied an enhanced Lee filter using a 3×3 window size to reduce noise. The images were processed using SNAP 7.0.3 software. A LiDAR DEM from the Spanish Geographic Institute, with an absolute vertical accuracy of 0.20 m, was used for the elevation measurements.

Based on the map of synthetic habitat units in Castañeda et al. (2020), three main habitat domains were delimited below the 995 m contour line: lakebed, perilacustrine fringe, and terraces. We assumed that these domains were associated with different backscatter coefficient responses. The σ^0values for each habitat domain were analysed using ArcGis v.10.5 and SPSS v.27 software. The central tendency of the σ^0value distribution in each domain was illustrated with boxplot graphs that included the 95% confidence intervals for the medians, the interquartile ranges, and extreme values.

Ancillary data included continuous records of lake water level and meteorological data on rain and wind speed registered for the lake at Los Picos weather station (Fig. 1). We followed the criteria of Díaz de Arcaya et al. (2005) to interpret the actual water extent of Gallocanta Lake.

RESULTS and DISCUSSION

Overall, the three domains showed that VH σ^0 was significantly lower than VV σ^0, and had a lower range (14.2 dB and 17.1 dB, respectively). The median VH σ^0 was the lowest for all the domains, at <-22 dB (Fig. 1), and therefore prevented any differentiation being made. The median VV σ^0showed significant seasonal differences for the lakebed domain. The accumulated rainfall (6 mm) in the 5 days preceding the dry season image (16-10-2019) could have increased the surface soil moisture and affected the radar response. The wind velocity was similar on the two dates studied, at about 5 m s^{-1}, which was strong enough to influence the $\sigma^0$of the open water surface (Martí et al., 2010) in the humid season (25-02- 2020). The backscatter characteristics of each domain is analyzed following.

Lakebed domain

The lakebed encompassed 979.8 ha and represented 46.8% of the study area. This domain included the current intermittently flooded area with a mean elevation of 991.7 m a.s.l. An almost flat bare bottom is expected in non-flood conditions, with a continuous sheet of open water during the flood season. If the lake water persists during the year, some aquatic vegetation may develop.

On average, the lakebed showed low backscatter and a significant seasonal difference (Fig. 1). The median σ^0 was -16.6 dB and -22.6 dB for the dry and flooded dates, respectively, in agreement with the lake water level of 12.5 cm and 61.5 cm. Based on our field experience, this dry season water height does not imply the occurrence of a continuous surface water across the lakebed. Under dry conditions, a low σ^0 value was expected for this domain, particularly as the preceding rainfall left the soil surface saturated in water.

In the flood season, the open water showed lower σ^0values than the water-saturated soil in the dry season, ranging from -6.6 to -27.5 dB. This was also observed by Martí et al. (2010) using Radarsat data in other intermittent wetlands. The bimodal distribution of the flood histogram (Fig. 2) showed that a section of the lakebed was affected by surface roughness, probably due to wind action. Meanwhile, the unimodal histogram of the dry season indicated a homogeneous response of the lakebed as viewed by Sentinel-1.

Perilacustrine Domain

The perilacustrine domain covered 725.2 ha, 34.6% of the study area, and had a mean elevation of 993.1 m a.s.l. This domain surrounded the lakebed (Fig.1A) with patches of herbaceous vegetation, varying according to its salinity and flood tolerance (Castañeda et al., 2020). Salt meadows on the wet saline soils and brackish wetlands covered 91.5% of the perilacustrine area. Locally, small patches of flooded reed beds or other dense perennial grasses were present.

The backscatter presented a wide range of values for the two seasons, even though the results produced unimodal σ0 distributions (Fig. 2). The σ0 medians were -16.8 dB and -

15.1 dB for the dry and flood seasons, respectively. The wide σ^0range in both seasons can be explained by the mixed response of the radar signal due to the diversity of herbaceous wetland vegetation (Martí et al., 2010) and the potential presence of water in low-lying areas.

The σ^0range in the flood season was the highest recorded from all the domains (Fig. 1A), from -26.2 dB to 5.1 dB. For the dry-season image, the σ^0range (-6.5 dB to -23.4 dB) may have been affected by the accumulated rainfall (6 mm). For the highly saline habitats, principally pioneering halophytes with low vegetation cover extended along a narrow fringe occupying 18% of this domain (Table 2, Castañeda et al., 2020), there were no backscatter differences compared to other herbaceous vegetation in less saline areas.

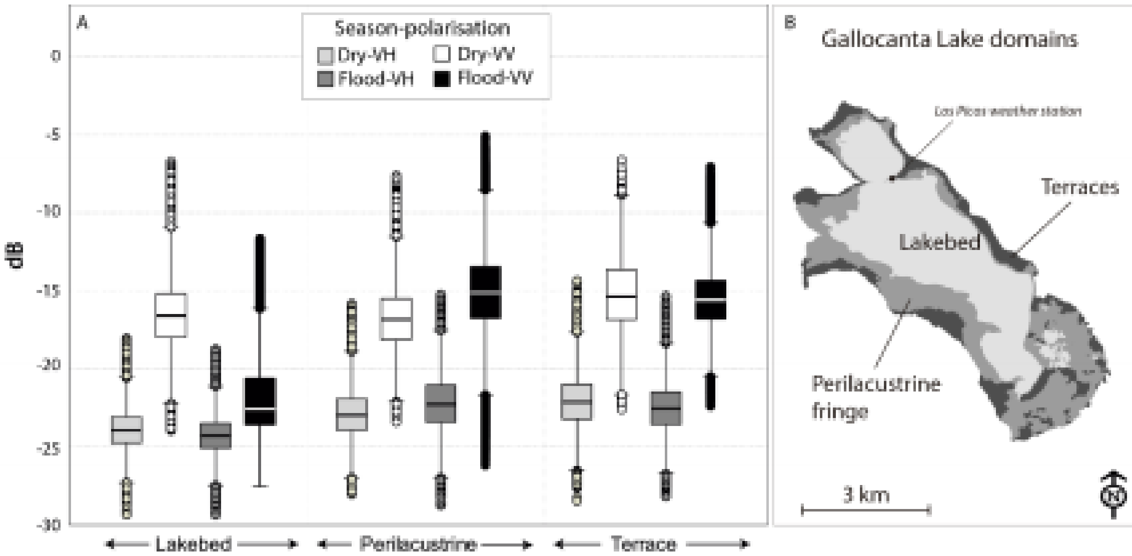

Fig.1.
A: Boxplot diagram showing the distribution of the backscatter values (dB) for each lake domain and season. B: Lake domains and weather station location.

Terraces Domain

The terraces, currently distributed along the external lake fringe (Fig. 1B), were ancient perilacustrine sedimentary plains. The terraces covered 387.9 ha, 18.5% of the study area, and had a mean elevation of 994.2 m a.s.l. They corresponded to crops and ruderal vegetation occupying the driest soils around the lake.

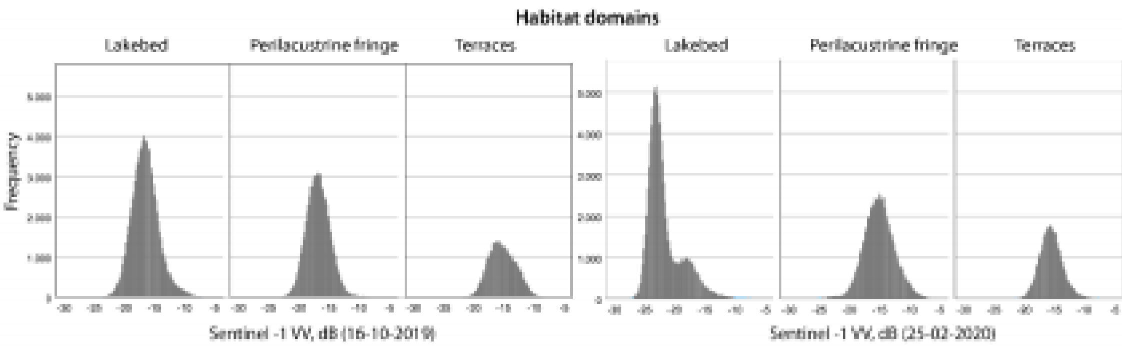

Fig.2. Sentinel-1 histograms of VV $\sigma^0$for the three lake domains.

The backscatter response was analogous in the two seasons, with similar σ^0 medians and ranges probably due to the similarly low level of cereal crop cover as this is typically sown in October. The σ^0 range of the terraces was the lowest of the three domains, and its unimodal distribution indicated a homogeneous response to Sentinel-1 on the two dates analysed.

CONCLUSIONS

The response of the habitats of Gallocanta Lake to Sentinel-1 VV backscatter showed a significative difference between dry and flood seasons only for the lakebed, identifying the presence of open water. The perilacustrine domain showed the widest backscatter range and a complex response due to various vegetation types and cover.

The habitats response to VV backscatter of Sentinel-1 could be complemented with a higher resolution of X-band radar images in order to discriminate different vegetation types. Moreover, future analysis should be applied to a series of images during various conditions of lake water level. Interpretation of backscatter should be based on simultaneous field data on water level and local conditions.

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Analysis of the changes in Lake Prespa and its surroundings over the last 4 decades by remote sensing methodology

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INTRODUCTION

The Ohrid-Prespa lake system (Fig.1) is the oldest and most diverse permanent lake system in Europe dating from the Pliocene with more than 1 Ma of existence. Its smaller component Lake Macro Prespa (thereafter called Prespa), located at the border of North Macedonia, Albania, and Greece (40º46´–41º00´N, 20º54´–21º07´E) is a tectonic lake situated at 849 m a.s.l the exact age of which is still uncertain; although the endemic faunal species it hosts, despite being fewer, have been suggested to be older than those in ancient Lake Ohrid (Karaman, 1971).

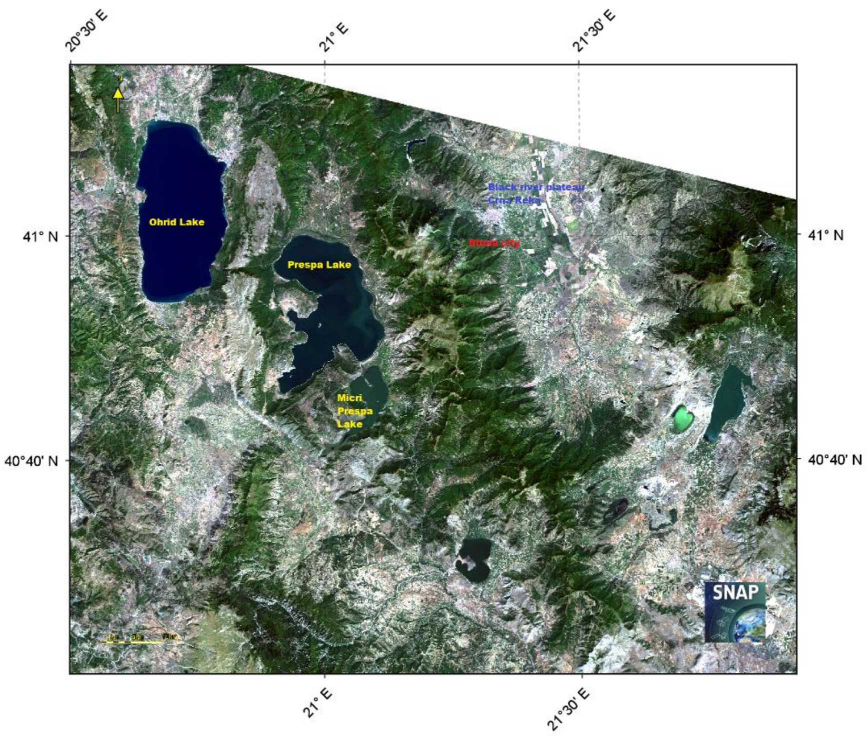

Fig.1. Geographical localization of the area of interest.

Lake Prespa is a relatively shallow lake with a water depth reported recently as 14m mean and 48m maximum. The water input, 16.92 m^3/s, is via river inflow and catchment runoff (56%), direct precipitation (35%), inflow (9%) from the nearby Lake Micri Prespa (shared by Greece and Albania), and groundwater input. The output is via evaporation (52%), water abstraction for irrigation (2%) and subsurface outflow through the karstic aquifers of Galichica Mountain (46%) that feeds lake Ohrid (Matzinger et al., 2006). The lake is highly sensitive to external impacts including climate change and has been suffering major water loss over past decades. Lake-level decline of almost 10m was documented between 1950 and 2009 due to restricted precipitation and increased water abstraction for irrigation (Popovska and Bonacci, 2007). The rate of decline has increased over the past decade and currently the water level is the lowest since records have been available (100 years).

AIM AND METHODS

This study describes the changes in the surface size of the lake and the vegetation/land use in the surrounding area in the period 1984-2020 using satellite images (remote sensing, Landsat 5 & 8 images by ESA and USGS). Satellite maps show the density of plant growth and can be used to quantify the amount and type of the vegetation present. One of the most used indices is calculated from multispectral information as the normalized ratio between the reflectance in red and near infrared bands and is called Normalized Difference Vegetation Index (NDVI) (Karnieli et al., 2010) which is used directly to characterize canopy growth or vigor as chlorophyll actively absorbs red and reflects near infrared light. NDVI values range from -1 to 1; negative values approaching -1 correspond to water, values close to zero (-0.1 to 0.1) correspond to barren areas of rock, sand, or snow, moderate values represent shrub and grassland (0.2 to 0.3), high values indicate temperate forests (0.6 to 0.8) while tropical rainforests are approaching 1.

RESULTS

Analysis of satellite images revealed that the lake lost 18,87 km^2 of surface in the period between 1984-2020 (Fig. 2) dropping from 273,38 km^2 (June 1984) to 254,51 km^2 (July, 2020)- in other words, the lake lost 6,9% of its size. The rate of water loss was greater in the period 1987-1993 and 1998-2004 while the surface of the lake has not varied in the last decade.

NDVI is a measure of greenness calculated from the visible and near-infrared light reflected by vegetation as green vegetation absorbs most of the visible light that hits it, and reflects a large proportion of the near-infrared light. Analysis of NDVI in the area surrounding lake Prespa revealed an increase in the mean NDVI values in the area over the period studied 1984-2020 (Fig. 3). In the studied area (app. 4950 km^2, Fig. 2B), the vegetated surface (NDVI > 0.13) increased from 78% in 1984 to 86% in 2020. In particular, the area with the highest vegetation intensity (NDVI > 0.45) has increased by 40%. This contributes on the one hand to the retention of runoff in the forest areas and on the other hand to water consumption in the irrigated areas.

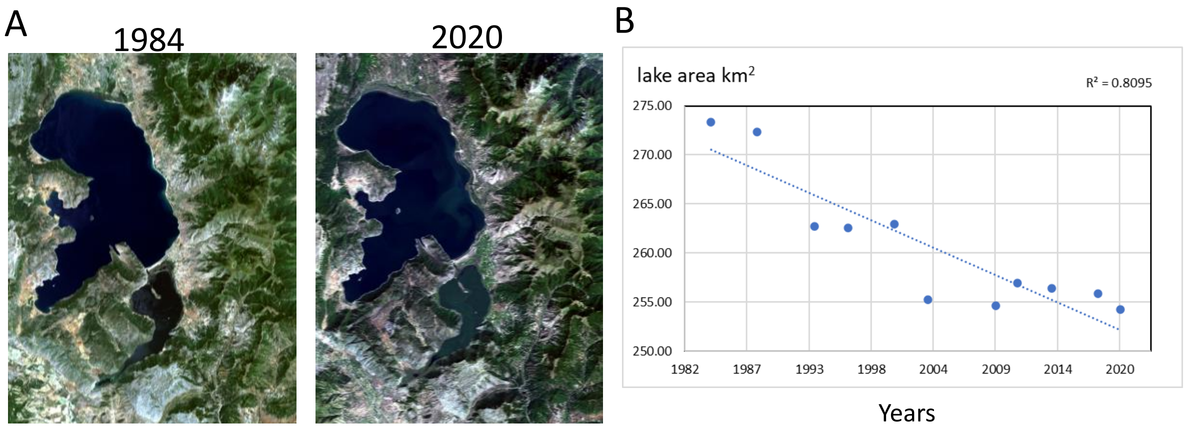

Figure 2: (A) Satellite images of Lake Prespa. (B). Calculation of the lake size in the period 1984-2020.

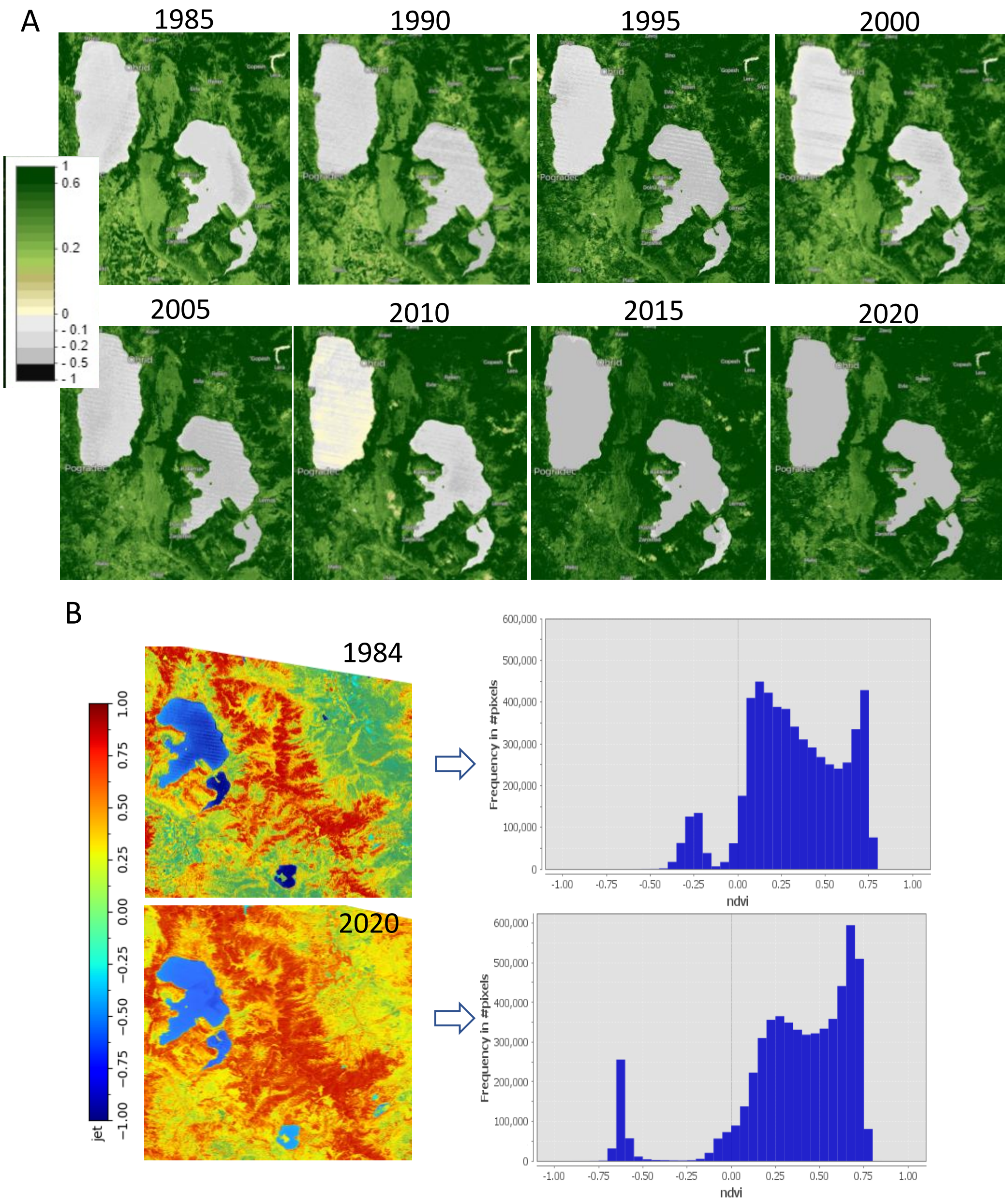

Figure 3: (A) Normalized Difference Vegetation Index (NDVI) in the period 1984-2020. Images were taken in the summer period (June-August) by Landsat-5 (1985-2010) and by Landsat-8 (2015 and 2020). (B) False color images for 1984 and 2020, and histograms showing color-NDVI values.

CONCLUSIONS

Lake Prespa has suffered a dramatic drop of the water level over the past decades. Remote sensing methodology is a useful way to study changes in Lake Prespa regarding size and changes in the vegetation in the surrounding area. This increase in the vegetation may be associated with enhanced irrigation in the surrounding area for agricultural use, a factor participating in the loss of water from the lake. The survival of the lake requires urgent measures, strong legislation and better cooperation among the three countries involved in management of the lake.

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Studying ponds ecosystems for climate Change mitigation: H2020 PONDERFUL project

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Freshwater ecosystems host some of the most diverse communities worldwide but are also among the most vulnerable to human-driven environmental change. In particular ponds (water bodies smaller than 5 ha) support 70% of the freshwater species pool in the European landscape, being the most numerous among freshwater ecosystems (100 times more ponds than lakes)[1]. Despite their abundance and key role as habitats for biodiversity and ecosystems functions and services, small ponds have been largely ignored by freshwater biologist and neglected by policy makers and management[2]. However, ponds may be particularly vulnerable to climate change and anthropogenic activities, such as land use intensification, being less buffered to other factors such as temperature increase, heat waves and changes in hydrology[3].

PONDERFUL (Pond Ecosystems for Resilient Future landscapes in changing climate) is a multidisciplinary EU Horizon 2020 project, involving 11 countries across Europe (and including Uruguay), which, will investigate the role of ponds and pondscapes (networks of ponds) in mitigating and adapting to climate change, protecting biodiversity and delivering ecosystems services over the next four years. The main objective is to study the responses of ponds biodiversity to environmental changes (including temperature changes, changes in hydroperiod and land use intensification), and quantify the relations between biodiversity, ecosystems services and climate changes developing scenarios for climate mitigation. To fulfil this objective, PONDERFUL will collect different parameters (e.g., pond metabolism and biodiversity) across a longitudinal transect from the south to the north of Europe and including Uruguay (Figure 1).

Here, I will introduce the aims of the PONDERFUL project and describe the role of ponds in the European landscapes, and how we could use ponds and pondscapes as a nature-based solution, developing practical tools for managements of pondscapes.

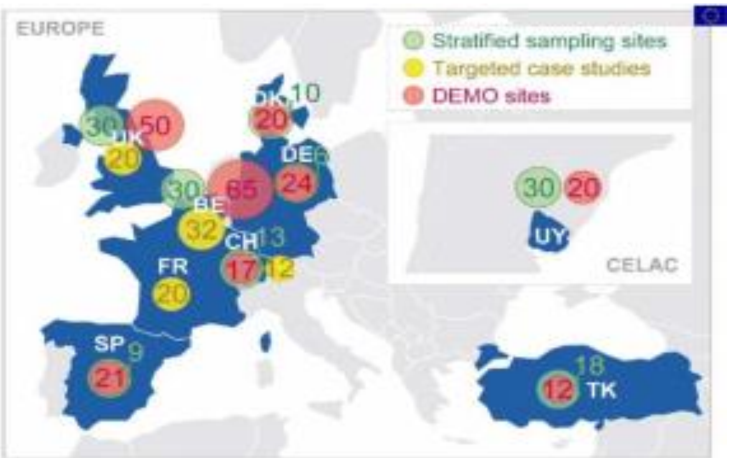

Figure 1: Number of ponds in the stratified sampling sites (always 30 ponds per country, including ponds to test hydroperiod effects),

targeted case studies sites (where in some ponds work has already started) and DEMO sites (which in some regions partially match the stratified sampling ponds).

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Developing DEMO-sites to highlight the efficiency of Pondscapes as Nature Based solution for adaptation and mitigation to changing climate (H2020 Ponderful project)

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One main aim of PONDERFUL is to provide evidence that ponds and pondscapes, as Nature Based Solution (NbS), can help society to mitigate and adapt to climate change, protect biodiversity and deliver ecosystem services. Indeed, ponds and pondscapes have already been highlighted as key-systems for biodiversity. Their services, however, also include impacts on greenhouse gases fluxes, water provision, flooding, extreme events control, regulation of water quality, learning, inspiration and recreation, which all impact human well-being.

A key outcome of PONDERFUL will be to provide the legislators, policy makers, practitioners, land managers, farmers, and SMEs with the guidance tools to support the establishment of a low-cost and highly effective approach for implementation of NbS based on ponds and pondscapes.

Here I will present a network of 15 DEMO-pondscapes that we co-develop with stakeholders throughout Europe and Uruguay. These pondscapes are representative of the diversity of NbS that can be easily implemented elsewhere. I will explain here how we are assessing their effectiveness for increasing resilience of biodiversity and ecosystem services to climate change, highlighting best practices to be promoted in Europe.

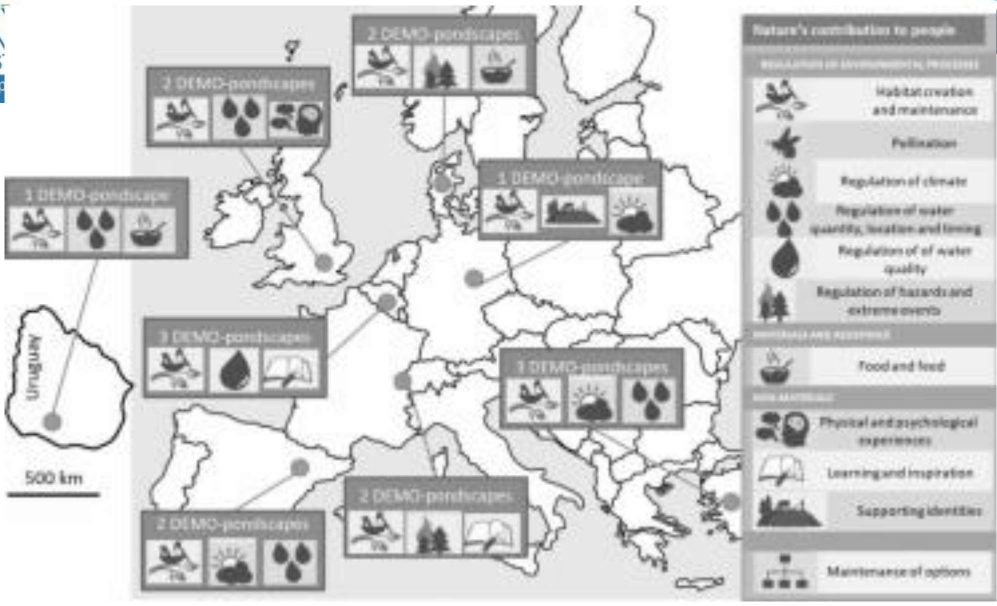

Fig. 1. Geographic location of the 8 DEMO site regions in Europe and Uruguay, where in total 15 DEMO-pondscapes will be set up, presenting examples of Ecosystem services and "Nature's Contributions to People" offered by the Nature Based Solutions implemented in various pondscapes. A list of investigated ES/NCPs is presented on the right side of the figure, following IPBES (2019).

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Keynote 5: Wet grassland restoration: principles, practice and prospects

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INTRODUCTION

Wet grasslands are biologically diverse wetlands dependent upon hydrological regime and vegetation management for their particular characteristics. These wetlands are defined by an abundance of grasses (or sedges), as well as periodic flooding with fresh or brackish water, or a high water table for some of the year, sufficient to influence the vegetation (Joyce et al., 2016). Many wet grasslands are “seminatural”, formed by partial drainage of other wetland types or forest clearance on floodplains but still largely comprised of characteristic native plant species. Wet grasslands provide multiple ecosystem services, including flood attenuation, groundwater recharge, carbon storage, nutrient removal, and aesthetic value.

It is likely that the wet grassland resource experienced losses of at least 80% during the 20th century, mostly due to drainage and agricultural changes. Agricultural intensification has destroyed or damaged most wet grasslands and those that remain are often threatened by neglect or abandonment (Joyce, 2014). Consequently, initiatives to restore wet grasslands have begun, especially in Europe. The aim of this presentation is to review restoration of wet grasslands in relation to common principles and key management practices, and to consider the prospects for wet grassland restoration in the face of emerging issues.

PRINCIPLES

Identifying the potential for wet grassland restoration depends upon an understanding of the properties of each site so that appropriate management decisions can be taken, particularly related to hydrology and vegetation. Making the most of any opportunity requires careful consideration and planning. Objectives should be defined based upon knowledge of the existing and potential value of the site. It is vital that restoration activities safeguard existing values and services rather than compromise them. For sites with existing values, such as residual diversity, altered or reinstated management may be sufficient to restore wet grassland communities. For other wet grasslands, additional intervention is required to ensure an adequate water regime and enhance biodiversity, such as deliberate species introductions (e.g. as seeds) to sites isolated from potential sources of colonizing species. Management undertaken should be subject to monitoring, which should be planned and adequately resourced at the outset. The ability to modify management in the light of results from monitoring is important if any unforeseen, undesirable effects are to be mitigated (Benstead et al., 1999).

PRACTICE

Hydrological management

Wetlands are defined by their hydrological regime so water management is critical for successful restoration of wet grasslands. Maintenance of natural hydrological processe should be a primary objective. For example, flooding can facilitate restoration by importing seeds or other propagules to wet grasslands. Where a wetland water regime has been lost, restoration of wet grassland usually relies on the establishment of suitable hydrological conditions, often by raising water levels. Research from southern England has shown that drained grassland plant communities are responsive to raised water levels and have potential for a rapid transition to wetland vegetation (Toogood and Joyce, 2009). Water regime, measured as duration of flooding, groundwater level and soil moisture, was significantly related to plant community variation. With increasing wetness, sites were characterised by more wetland plants such as sedges, helophytes and hydrophytes, and species with a stress-tolerating competitive strategy. All sites showed considerable annual dynamics, especially those with substantially raised water levels. Thus, creation or restoration of wet grasslands by (re)wetting is feasible but challenging due to the high dynamism of wetland plant communities.

Vegetation management

The restoration of abandoned wet grasslands through vegetation management may be more feasible compared to those severely degraded by agricultural intensification or eutrophication, where successful restoration has proven difficult. Following abandonment, the hydroperiod generally remains suitable for wet grassland biota, so restoration of abandoned grasslands most often replicates former agricultural vegetation management, such as cutting for hay and/or grazing, in order to reverse succession. However, it is possible to reduce nutrient levels to restore wet grasslands by topsoil removal, and hay cutting and harvesting may be more effective than grazing if a key concern is nutrient removal. Research by Berg et al. (2012) on Estonian wet grasslands found that plant communities with a variable hydrological regime are more responsive to reinstated cutting management than those with stable water levels because the variable hydroperiod creates a dynamic environment favouring adaptable species.

PROSPECTS

Some of the key issues impacting the prospects for wet grassland restoration are:

Climate change: climate change is predicted to alter wet grassland hydrology, especially through warming, seasonal precipitation variability, and the severity of extreme events such as droughts and floods. Extreme storm or flood events will favour ruderal plant species able to respond rapidly to environmental change. In some regions, wet grasslands may dry out during heatwaves and drought. Restoration plans will need to adapt practices to mitigate climate change, with greater emphasis on water maintenance, flexible management, monitoring, and restoration of resilient wet grasslands.

Carbon storage: opportunities for wet grassland creation and restoration to enhance carbon storage will increase, although managing such sites may involve compromises with other services, such as agricultural production.

Wilding and nature-based management: opportunities for less intensive management of wet grasslands, for example through natural water regimes and extensive grazing by roaming animals, are possible through (re)wilding initiatives. Policies promoting nature-based solutions should benefit the restoration of wet grasslands for ecosystem services such as flood storage, removal of pollutants and nutrients, and for public access.

Invasive species: European wet grasslands have not generally been severely impacted by invasive non-native species but in North America problems with invasive species in restored wet grasslands are common. Invasive species could benefit from climate warming scenarios, facilitated by disturbances such as droughts, floods, and possibly wildfires.

CONCLUSIONS

Wet grasslands are biologically diverse wetlands defined by their hydrological regime and vegetation management that have suffered major losses and degradation, largely due to agricultural intensification and abandonment. It is therefore important to plan and implement wet grassland restoration, focussing on the maintenance of an appropriate hydroperiod and reinstating suitable vegetation management. Research shows that restoration success is dependent upon initial site conditions, and that while sites can be rehabilitated, fully successful restoration to a previous condition is elusive. This is partly due to the dynamic nature of wet grassland plant communities and their variable responses to water regimes and management practices. There are opportunities for more restoration of wet grasslands through emerging environmental policies but climate change and invasive species represent growing concerns.

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Keynote 6: About smaller and bigger kidneys: riparian zones as nutrient buffers in Denmark

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INTRODUCTION

In Denmark, measures reducing nitrogen (N) and phosphorus (P) losses from fields are divided into two main categories “source mitigation measures”, e.g. catch crops and fertiliser norms as well as set-a-side and afforestation, and “nutrient transport mitigation measures”, e.g. restored wetlands and a number of drainage mitigation measures (Figure 1). This paper deals with nutrient transport mitigation measures to reduce diffuse pollution from agriculture. It treats already approved measures, such as restoration of riparian wetlands, larger lowlands areas including fens and swamps, re-establishment of shallow lakes, constructed wetlands (surface flow and subsurface flow), as well as drainage mitigation measures not yet approved and still under development such as integrated buffer zones, saturated buffer zones and controlled drainage.

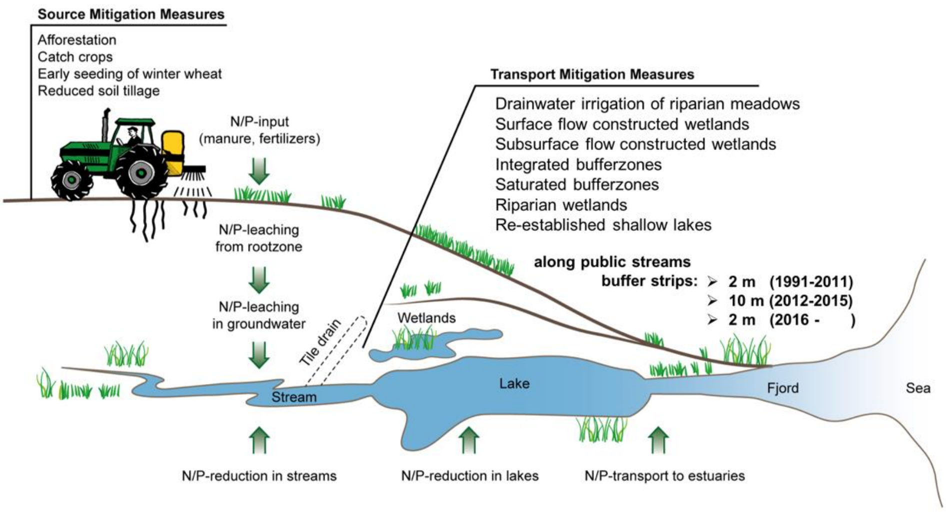

Fig. 1. Nutrient reduction efforts in Denmark since 1985.

METHODS

New nutrient transport mitigation measures cannot be implemented in Denmark until after completion of a series of steps. Whenever a new measure is proposed for use by Danish farmer advisors, it must be scientifically tested and thoroughly described. Thereafter, guidelines and national maps showing how and where to implement the measures must be made. A web-based support system for the funding of nutrient transport mitigation measures, including construction criteria, guidelines and maps, is run by the Danish Ministry of Environment and Food (lbst.dk/tilskudsguide).

RESULTS and DISCUSSION

Wetland restoration measures have proved to be efficient at removing N, whereas the results regarding P are more variable; in fact, some sites have been observed to act as P sources, especially in the first years following rewetting (Walton et al., 2020). Overall nutrient removal rates and efficiency vary strongly for all of the studied nutrient transport mitigation measures (Table 1). It is important to note that this variation not only reflects differences in efficiency of the mitigation measures but also differences in nutrient loading and local characteristics of the sites used for implementation (e.g. soil type, vegetation, climate) (Carstensen et al. 2020).

Table 1. Overview of absolute and relative nutrient removal efficiency (mean ± sd) of Danish nutrient transport mitigation measures (from Hoffmann et al. 2020)

	Sites (n)	Years	Removal rate (kg ha^{-1} y^{-1})		Removal efficiency (%)	
			TN	TP	TN	TP
Restored riparian wetlands	9	9	144±73	3±5	37±31	12±15
Restored shallow lakes	11	12	159±53	4±6	45±21	-2±83
Restored swamps and fens	5	5	209±77	2±3	44±12	11±26
Drain water irrigation	10	10	139±91	-0.3±0.3	45±22	-51±49
Surface flow constr.wetland	13	44	472±372	31±26	23±10	45±20
Subsurface flow constr. wetland	3	15	7771±241	34±6	50±13	12±4
Controlled drainage	4	8	8.8±6.5	2.2±2.4	33±13	5±29
Integrated buffer zones	3	6	1661±605	17±15	45±12	29±60

CONCLUSIONS

The Danish strategy to mitigate agricultural nutrient losses has resulted in a substantial decrease in the nutrient export to fresh waters. Yet, more efforts are still required to reach the “good ecological status” stipulated in the EU Water Framework Directive. Furthermore it is recognized that other aspects, for example, biodiversity or greenhouse gas emissions, need to be included in montirong schemes to support the implementation of mitigation measures.

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Impacts of climate Change on Wetland Plant Communities: A Mesocosm Study

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INTRODUCTION

Coastal wetlands are considered valuable ecosystems for their biodiversity and the wide range of ecosystem services they supply, including sediment retention, water filtration, nutrient recycling, flood regulation, carbon sequestration, recreational activities and maintenance of productive coastal fisheries (Engle 2011, Villoslada et al. 2019, Lima et al. 2020, Ward 2020a). However, coastal wetlands worldwide are also subject to various impacts resulting from natural and anthropogenic drivers, such as urbanization and residential developments, conversion to agricultural land and climate change related impacts such as sea level rise, changes in precipitation, inundation, changes in salinity and erosion (Ward 2020b).
The consequences of climate change over recent decades are evident in the Baltic Sea region, with modifications in sea water circulation, temperature and salinity (Omstedt et al., 2004; Räisänen, 2017). In the Baltic Sea, both salinity and water levels have been shown to have a strong influence on coastal plant community composition (Berg et al., 2011; Ward et al., 2016a). In this regard, the Baltic coastal wetlands are expected to be altered due to climate change (Ward et al., 2016b).
Due to the high importance of coastal wetlands and the impacts of climate change on this ecosystem, it is essential to determine how future conditions will influence coastal plant community functioning. Previous studies in Estonia showed changes on the coastal meadows related to microtopography (Ward et al., 2016b), impacts of management and grazing (Berg et al., 2011), impacts on coastal ecosystems due climate change focused on precipitation and sea level rise (Kont et al., 1997; Ward et al. 2016b). The aim of this study is to evaluate the effects of altered water level and salinity conditions on three different wetland communities in Estonia using medium-term (3 yr.) mesocosm experiments to evaluate the changes in the plant communities' composition under different treatments.

METHODS

Study site

The Baltic Sea is one of the largest brackish water bodies due to its relative isolation as a consequence of the narrow connection with the Atlantic Ocean through the Danish Straits (Kont et al., 2003). It is strongly influenced by large-scale atmospheric circulation, hydrological process and restricted water exchange in its entrance (Graham et al., 2007). Salinity within the Baltic Sea is maintained by a pattern of stratification (Stigebrandt,2001). The outflow of low salinity water in the surface and the inflow of higher salinity water at depth maintain the upper layer salinity at about 6–8 psu (practical salinity unit) around Estonia and a more saline deep

water layer with about 10–14 psu (von Storch & Omstedt, 2008), although this varies geographically.
The hydrological regime is characterized by very low tide (~0.02 m range), and flooding is predominantly driven by atmospheric pressure and fluctuating meteorological conditions across the North Atlantic and Fennoscandia (Suursaar & Sooäär, 2007). As a result of this, the rate and magnitude of inundation is irregular throughout the coastal landscape (Rivis et al., 2016). Recent estimates of relative sea level rise from three tide gauges along the Estonian coast are: 1.5–1.7 mm yr^{-1} at Tallinn, 1.7–2.1 mm yr^{-1} from Narva-Jõesuu and 2.3–2.7 mm yr−1 at Pärnu (Ward et al., 2014).

Experimental design
Future scenarios of Estonian coastal wetlands were evaluated using a three-year mesocosm experiment (2018-2020) simulating altered environmental conditions of water level and salinity. The response of three characteristic plant communities (Open Pioneer- OP, Lower Shore- LS and Upper Shore- US) were assessed in terms of changes in species composition through time. The experiment included 45 mesocosms, 15 per community with 5 treatments (3 replicates per treatment) with control, altered water level and salinity (increased and decreased levels from control).
The responses of communities were evaluated through abundance of plant species present by area of ground cover during the growing period (from April to September) using a 50 cm^2 permanent graduated quadrat.

Statistical analysis
In order to analyse the differences among treatments in plant communities through time, Permutational Multivariate analysis of variance (PERMANOVA) and species contribution were performed using R software (R version 4.0.3). Bray-Curtis dissimilarity was calculated from species importance values. When a significant ($p < 0.05$) difference between the treatments was detected, a dissimilarity percentage analysis (SIMPER) was used to reveal the contribution of species for the differences between treatments.

RESULTS and DISCUSSION

All factors, year and treatment, influenced plant communities composition (Table 1). Based on factor effects, the year explained the most of variation for Open Pioneer and Lower Shore communities, followed by treatment. For Upper Shore, treatment explained the most of variation (treatment $R^2 = 0.05$; year $R^2 = 0.03$; Table 1).

The results showed changes in plant species contribution (Table 2). For instance, in US, *Poa angustifolia* (*P. angustifolia*) responded positively to increased water level and salinity compared to control, while *Carex nigra* (*C. nigra*) showed an increase of importance in decreased water conditions. *Festuca rubra* (*F. rubra*) responded with higher importance contribution on decreased salinity conditions. These results demonstrate that, even in a relatively short time period, the communities showed significant changes under different drives. Other studies have found changes in species composition as a response to altered salinity (Janousek et al. 2013) and hydrology (Brotherton et al. 2019).

Table 1. Effects of experimental factors on plant communities composition.

	Source of variation	MeanSqs	F. model	R2	*p* (perm)
OP	Treatment	25.7	360.3	0.15	<0.001
	Year	63.03	883	0.19	<0.001
	Residuals	0.07		0.53	
LS	Treatment	11.7	112.21	0.07	<0.001
	Year	30.04	228.14	0.09	<0.001
	Residuals	0.1		0.79	
US	Treatment	4.23	67.8	0.05	<0.001
	Year	4.99	680.03	0.03	<0.001
	Residuals	0.06		0.8	

Table 2. Species contribution (%) for 2018 and 2020 in Upper Shore community.

		Control vs T1 (%)	Control vs T2 (%)	Control vs T3 (%)	Control vs T4 (%)
2018	*C. nigra*	65	59	63	-
	F. rubra	-	-	-	-
	P. angustifolia	-	-	-	-
	Bare ground	-	-	-	-
	Litter	38	33	37	37
2020	*C. nigra*	-	77	65	-
	F. rubra	66	-	-	77
	P. angustifolia	80	69	76	-
	Bare ground	52	61	54	67
	Litter	31	67	77	-

CONCLUSIONS

Altered water level and salinity changed the communities composition. Our results suggest that climate change, and consequently changes in water level and salinity in the Baltic Sea, can have important impacts on coastal meadows in the region, highlighting the importance of conservation for these areas due to their important ecosystem services and value for biodiversity support.

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Do common reed leaves incrustations affect litter decomposition?

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INTRODUCTION

Wetlands with prevailing helophyte vegetation exhibit high primary productivity of biomass, the majority of which is subjected to decomposition (Longhi et al., 2008). The prediction of fate and persistence of plant litter in wetlands is not easily predictable, as plant material and conditions for the decomposition process may be highly heterogenous. It was shown that decomposition rate is strongly affected by element composition of plant litter, including phosphorus (P) (Saggar et al., 1998) and silicon (Si) contents (Schaller and Struyf, 2013), and also by water and temperature. Si is the most important plant biomineral. Having the ability to alleviate negative effects of unfavourable abiotic and biotic factors, such as drought, extreme temperatures, toxic metals, high salinity and radiation, and pathogen attacks (de Camargo et al., 2019), it is a beneficial element for plants, especially for grasses. Silica bodies (also known as phytoliths) are deposits in plant tissues, where they function as structural support, as an alternative to lignin, and as defence against herbivores (Strömberg et al., 2016). However, these incrustations may affect the colonisation of plant litter with microbes.

Lake Cerknica is an intermittent wetland with pronounced water level fluctuations. The vegetation of Lake Cerknica is characterised by wetland communities, including Phragmitetum australis (Martinčič and Leskovar, 2003), which contribute significantly to primary production and consequently to the production of litter in the ecosystem (Dolinar et al., 2016). Even though fungal taxa of the *Phragmites australis* canopy and litter have already been identified, their link to plant litter decomposition is still unclear.

The aims of the present study were to compare the decomposition rate of the *P. australis* leaf litter of two different leaf ages and from two different locations at Lake Cerknica with different leaf Si contents, exposed to two different conditions, flooded and dry. In addition, we wanted to explore fungal communities of these samples prior to and after the decomposition process.

METHODS

The decomposition experiment was performed using the litterbag method. In September, litterbags with leaf litter of different age (lower and upper leaves) from two locations at Lake Cerknica (Zadnji Kraj and Gorenje Jezero) were placed at two microlocations at Gorenje Jezero. These two microlocations differed by the extent of water level fluctuations from almost completely dry to almost always submerged. The litter bags were collected after 45 days of exposure and the remaining plant material was used to determine element composition and fungal colonisation. To compare decomposition of leaf material from different locations, exposed to different conditions, we calculated the leaf litter decomposition rate.

Element composition of plant material prior to and after exposure was determined using X-ray fluorescence spectrometry (XRF). NextGen sequencing approach will be applied to characterise fungal communities on the initial and decomposed material.

RESULTS and DISCUSSION

XRF analysis of the plant litter revealed that leaf samples from Zadnji Kraj contained significantly higher amounts of Si, where lower leaves contained less Si in comparison to the upper leaves (Table 1). This is possibly related to transpiration flow (Grašič et al., 2019) that is expected to be higher in upper leaves, possibly due to specific environmental parameters. We also determined somewhat higher contents of P and some other elements in the leaves from Gorenje Jezero, which might be related to the permanent presence of nutrient-rich water at this microlocation, as shown by long-term monitoring (Gaberščik and Urbanc-Berčič, 2003). The level of titanium (Ti), which is also beneficial for plants, was higher in the leaves from Zadnji Kraj, and was positively related to leaf Si content (Pearson correlation coefficient, 0.732; $p < 0.01$).

Table 1. Element composition of the *Phragmites australis* leaves of different age and from different locations prior to the experiment

	Gorenje Jezero, upper leaves	**Gorenje Jezero, lower leaves**	**Zadnji Kraj, upper leaves**	**Zadnji Kraj, lower leaves**
Si	3.12 ± 0.61^{a}	2.84 ± 0.43^{a}	6.12 ± 0.82^{c}	4.98 ± 0.86^{b}
P	0.17 ± 0.04^{b}	0.17 ± 0.03^{b}	0.12 ± 0.02^{a}	0.12 ± 0.04^{a}
S	0.35 ± 0.06^{b}	0.30 ± 0.08^{ab}	0.32 ± 0.08^{ab}	0.23 ± 0.06^{a}
Cl	0.36 ± 0.24^{b}	0.16 ± 0.18^{a}	0.04 ± 0.04^{a}	0.02 ± 0.01^{a}
K	0.20 ± 0.06^{a}	0.29 ± 0.09^{b}	0.16 ± 0.06^{a}	0.17 ± 0.02^{a}
Ca	1.83 ± 0.16^{b}	1.51 ± 0.25^{a}	1.42 ± 0.22^{a}	1.50 ± 0.21^{a}
Ti	0.0010 ± 0.0002^{a}	0.0011 ± 0.0004^{a}	0.0018 ± 0.0003^{b}	0.0018 ± 0.0004^{b}

Decomposition rate was much lower in the dry microlocation in comparison to the wet microlocation, and was somewhat higher for upper leaves. In spite of the great differences in Si content of the leaf litter from Zadnji Kraj and Gorenje Jezero, no differences between these samples were observed in decomposition rate (Figure 1).

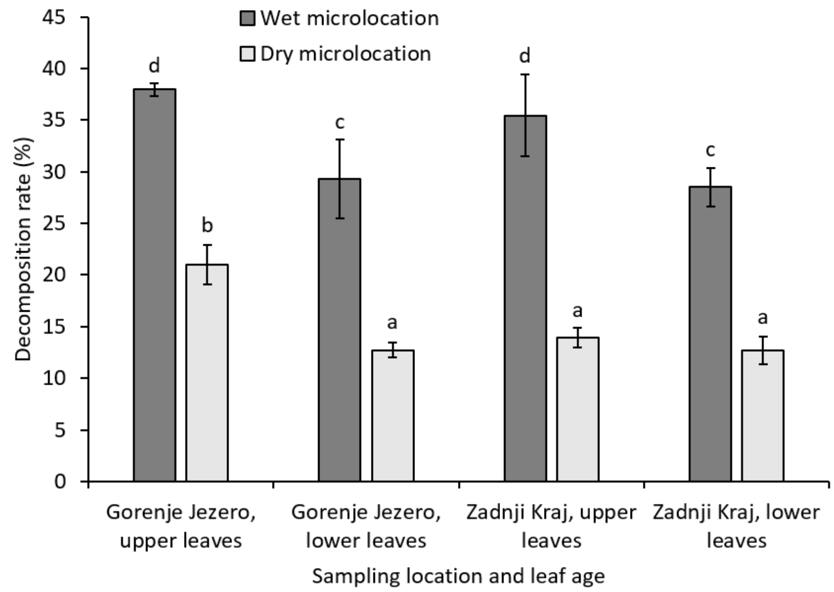

Fig. 1.

Decomposition rate of Phragmites australis leaves of different age and from different locations at the wet and dry microlocations

The decomposition rate of leaf litter was not related to leaf Si content (Table 2). Schaller and Struyf (2013) reported that carbon turnover during decomposition was strongly positively influenced by litter Si content. In our study, decomposition rate was positively related to leaf P content at the dry, but not wet microlocation. The absence of this correlation at the flooded microlocation was possibly the consequence of the continuous presence of water that may flush P, which is being released from the decomposing tissues. Different studies showed contrasting results about the effect of P on the decomposing material. Saggar et al. (1998) showed that the rates of decomposition were related to plant P content, while this was not the case in the study of Jalali and Rajnbar (2009).

Table 2. Pearson's correlations coefficients showing the relationships between *Phragmites australis* leaf element contents and leaf decomposition rate at the wet and dry microlocations, and for the both microlocations combined

Leaf element content (% DM)	**Leaf decomposition rate/microlocation**		
	Dry	**Wet**	**Combined**
Ti	-0.439*	-0.088	-0.171
Si	-0.305	0.112	-0.099
P	0.445*	0.084	0.171
S	0.634**	0.669**	0.294*
Cl	0.820**	0.479*	0.300*
K	-0.139	-0.338	-0.070
Ca	0.865**	0.477*	0.283*

*, $p \leq 0.05$; **, $p \leq 0.01$

CONCLUSIONS

The results of our study indicated that decomposition rate differs significantly according to the environmental conditions during the decomposition process. The amount of Si in leaf litter did not affect decomposition rate. However, the effect of leaf P was significant at the dry microlocation. Ongoing molecular analyses of fungal communities will provide additional information on these relations.

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Eutrophication trend evaluation in a Greek lake, using remote sensing analysis

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INTRODUCTION

Eutrophication has been one of the main threats to water quality in lakes and reservoirs since 1960. Despite the amount of research conducted over the past six decades, eutrophication remains a significant concern worldwide (Smith and Schindler, 2009). Agricultural and municipal sewage and runoff from fertilizers are the critical factors for excessive nutrient loading to aquatic ecosystems. In the last two decades, the amplification in nutrient loading (Le Moal et al., 2019), has emerged the need for Water Bodies (WB) quality status control.

The increasing frequency of the presence of eutrophic ecosystems is also linked to climate change. The European Member States have recognized the need to preserve, protect and restore water bodies' quality status. In this scope, the Water Framework Directive (WFD) 2000/60 was obliged to be enforced for all European Water Bodies (Zacharias et al., 2020), despite the disputes for increased expenses due to the implementation of the project (Prato et al., 2014).

Apart from the restoration, maintaining biota/biodiversity in aquatic ecosystems controls water quality pollution. After all, monitoring WB quality is a legislative commitment for the European Union (EC, 2003). Today, the most cost effective way to monitor and control the WB quality is through remote sensing. The remote sensing analysis provides the possibility of simultaneous monitoring of numerous WB in near real-time and minimal cost. Chartography of WB water quality is an essential tool in identifying the mechanisms that affect and act in the aquatic ecosystem (Poschlod and Braun-Reichert, 2017) and reflect climate change.

The purpose of this study is the assessment of eutrophication status and trend through satellite imageries. The analysis used is widely known as phenology. Phenology estimates chlorophyll-a values, assess the WB trophic state and predicts the evolution of the phenomenon.

MATERIALS AND METHODS

Study Area

The study area is Lake Orestiada, i.e., the lake of Kastoria (Figure 1). Lake Orestiada is one of the ten largest lakes in Greece, with an area of 28-30 km^2. The maximum depth of the lake reaches 9.1 m, while the average depth is 4.4 m, according to Orestiada's Water Management Body. The lake's required georeferenced area was collected from GEODATA.gov, supported by the Ministry of Productive Reconstruction, Environment, and Energy of Greece (https://geodata.gov.gr/el/dataset/limnes-elladas). GEODATA.gov includes open-source data for the country of Greece. The shapefile used in our research, has been uploaded in November 2015.

Lake Orestiada is of high environmental importance and is included within the "Natura 2000" network (Skoulikidis, 2009) (freely available also https://natura2000.eea.europa.eu/). Lake Orestiada receives many pollutants such as wastewater, agrochemical effluents and overpopulation along the coast (Matzafleri et al., 2013).

Fig. 2. Orestiada: The lake of Kastoria City in North Western Greece.

In situ biological measurements

After 2015, as imposed by the WFD, Greece submitted the first report on WB quality status. Biological, physicochemical, and other measured characteristics were publicly available through the Hellenic Center for Habitats-Wetlands (http://biodiversity-info.gr/index.php/el/lakes-data#!IMGP4731). The results published are for four years (from 2013 up to 2016), and they classify the trophic state of Lake Orestiada as "Eutrophic". There has been noticed a high variance of chlorophyll values. Average chlorophyll values are 43.07μg/l, whereas they range from 7.56μg/l and may rise to 283μg/l.

Satellite Data

The data used in this study includes eighteen and a half years (more specifically from July 2002 to December 2020) of 8-day composites with 250 m spatial resolution. The products selected were Surface Reflectance [R_{rs}] imagery acquired from MODIS level 3 products of full resolution (available with no cost from oceancolor.gsfc.nasa.gov). The data was downloaded through an R numerical code, who was appropriately designed to collect the tile including the Lake of Orestiada (185_032). This type of data has been used to create chlorophyll-a time series, and asses bloom events in many different regions throughout the world.

Analysis

834 georeferenced images had been downloaded, applying RStudio numerical algorithm, without manually omitting images of high cloud coverage for the tile selected. After the data collection, the lake's Area of Interest (AOI) was selected and isolated. Then, because the spectral channels 3 to 7 are of 500 m resolution, they were converted to 250 m so that all bands have a common scale (of 250 m, for this study). At this point, all the downloaded spectral channels were incorporated into a single new image.

The satellite images' Rrs values are then corrected to eliminate possible errors of the measurements, deriving from aerosols and atmospheric particles. Troubleshooting is achieved by applying statistical data processing and calculating the dataset's principal components, each for every channel, every 8-day period. The extreme values are first replaced with NA (Not Available). The value of the standard deviation is extremely large as it has been affected by the species of existing extreme values. Therefore, this control is sufficient to reject the extreme values. NAs are also attributed to negative values, since negative surface reflectance values do not apply in nature but are instead an error of the sensor used. The new principal components of our time-series are recalculated.

The algorithm of chlorophyll is calculated as suggested by NASA's Ocean Color Webpage (https://oceancolor.gsfc.nasa.gov/atbd/chlor_a/). Three channels are needed: blue, green, and red which corresponds to MODIS Aqua to wavelengths 469, 555, and 655, accordingly. Two algorithms were constructed, i) the Color Index algorithm as described in Hu et al. (Hu et al., 2012), and ii) the OCx algorithm as described in O'Reilly et al.(O'Reilly et al., 2000). For the MODIS sensor, OC3M O'Reilly coefficients were used. Chlorophyll algorithm refinements from the modifications of thresholds were also used to select the Color Index algorithm or the OC3M appropriately, as described in Hu et al.(Hu et al., 2019).

RESULTS and DISCUSSION

Through the study of the of the eight days values (not straight line) and the trendline of the time series presented (straight line), it has been found that in recent years the concentration of chlorophyll-a in the lake's water has shown increasing tendencies over the years (Figure 2). It was also found that the values of chlorophyll-a in water are determined not only by the seasonality of the phenomenon but also by other factors that affect eutrophication (such as the nutrients of the effluents), which differ from season to season, thus giving the maximum value for each year at a different time. According to Krug et al., multiple phytoplankton blooms in an area is something that is often observed (Krug et al., 2018)

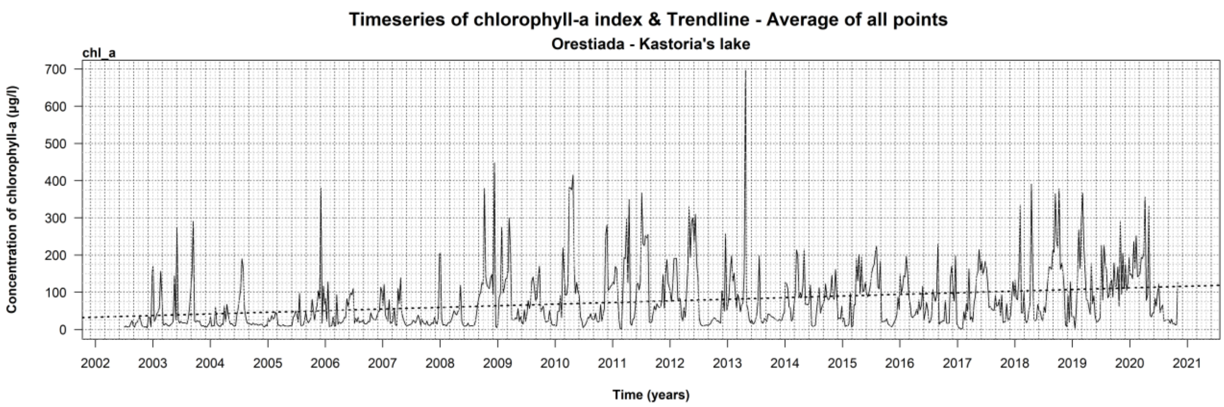

Fig. 3. Chlorophyll-a timeseries values after correction (straight line) and trendline (dotted line)

CONCLUSIONS

An increasing trend in chlorophyll-a concentration in the last years, can result in a change of the trophic status of Orestiada lake from eutrophic to hypertrophic. Over the years the phenomenon of eutrophication in Lake Orestiada was deteriorated considerably as the average values of chlorophyll-a (~ 100μg / l) and the extreme maximum values (~ 350μg / l) of the lake for the most recent years, range at consistently high levels, compared to previous years. This fact is due to the intense anthropogenic activities in the area such as wastewater, agrochemical effluents and overpopulation.

Additional field measurements can introduce the necessary data to validate the algorithms from the satellite data. To date, the sporadic in situ measurements, do not provide the ability to assess the evolution of chlorophyll in time. Remote sensing analysis is able to provide quality status analysis supplementary to situ measurements.

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Implementing surface flow constructed wetlands in Denmark using a new layout design

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INTRODUCTION

Nutrient enrichment can have detrimental effects on aquatic ecosystems, as excess nitrogen (N) and phosphorus (P) concentrations can lead to eutrophication, biodiversity loss, hypoxia, and loss of ecosystem services (Smith et al., 1999). Agriculture is one of the major sources of N and P to aquatic ecosystems worldwide as well as within the European Union. To mitigate these detrimental effects the Water Framework Directive 2000/60/EC by the European Union has implemented several environmental policies to protect freshwater resources (European Commission, 2000).

Denmark is an intensely cultivated country with approximately 62% of the land area being agricultural land (Danmarks Statistik, 2017). Furthermore, more than 50% of the agricultural land in Denmark is drained (Møller et al., 2018). Nitrogen passing through the subsurface drains is prevented from undergoing the natural N removal mechanism in soils (BlicherMathiesen et al., 2019), which underlines the need for implementation of mitigation measures targeting agricultural drainage water.

In 2015 the Agreement on the Food and Agriculture Package was implemented to allow Danish farmers to fertilize to economic optimum, breaking with nearly two decades of suboptimal fertilization. To comply with the EU directives while maintaining a high agricultural production a new nutrient management plan was implemented. This initiated a new era of management of agricultural nutrient losses in Denmark with emphasis on local environmental needs and mitigation measures. Several nutrient transport mitigations measures have currently been implemented or are undergoing development (Hoffmann et al., 2020). Surface flow constructed wetlands (CWs) are a popular best management practice when treating agricultural drainage water (Tanner et al., 2005). In 2010 the first two CWs were constructed in Denmark to investigate performance under Danish conditions using a new layout design. Experienced gained from these have been used to implement CWs in Denmark on a broad scale.

This study presents the design and implementation of CWs in Denmark and discusses their performances in general while using data from two CWs to illustrate performance over time.

METHODS

The CWs in Denmark follow the same design consisting of two components: (1) a sedimentation pond with a depth of 1 m and (2) the CW which consist of three deep open surface basins separated by two shallow zones with vegetation (Fig.1.). The sedimentation pond receives drainage water directly from agricultural fields. The deep basins of the CWs are 1 m deep to optimize the hydraulic retention time, while shallow zones of 0.3 m are designed to sustain vegetation and a continuous input of carbon.

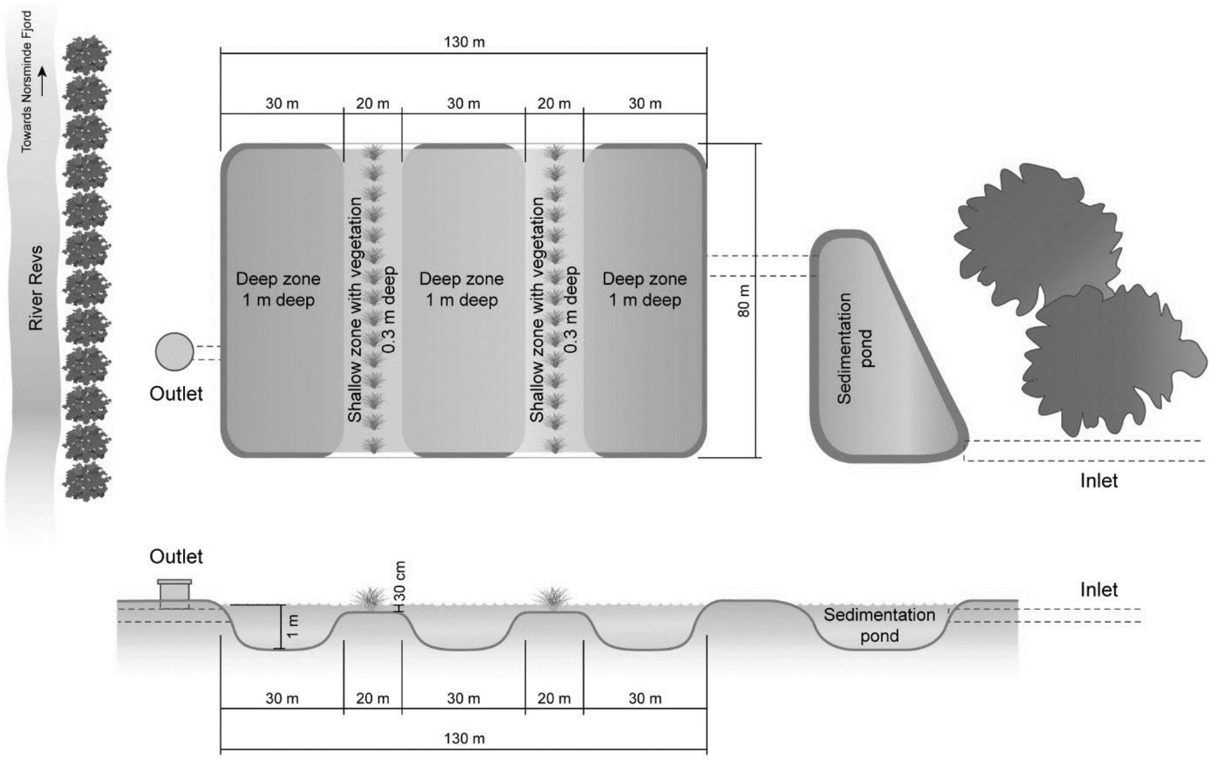

Fig. 1. Principle sketch of a surface flow constructed wetland consisting of a sedimentation basin and a wetland with a sequence of deep (~1m) and shallow (~0.3 m) zones.

The CWs at Fillerup and Fensholt are both constructed near Odder, Denmark. Fillerup was constructed in 2010 with a catchment area of 38 ha agricultural field. The CW is 0.298 ha in size and constitutes 0.78% of the drained area. The CW at Fensholt was constructed in 2015 and receives drainage water from 33 ha. It constitutes 0.74% of the drained area with a size of 0.245 ha.

Water samples were collected using ISCO samplers and all analyses comply with European standards. The N removal and P retention is evaluated using a mass balance approach.

RESULTS and DISCUSSION

In general, CWs in Denmark have shown to be effective for both N removal and P retention with mean removal rates of 472 and 31 kg ha-^{1} y-^{1} and mean removal efficiencies of 23 and 45%, respectively (Table 1). Nitrogen removal in CWs occurs through denitrification, while the P retention occurs mainly due to sedimentation of particulate P (Mendes et al., 2018). The efficiency of nitrogen removal shows strong seasonal variation, due to variations in hydraulic retention time (HRT) and temperature.

Table 1. Overview of absolute and relative nutrient removal efficiency (mean ± sd) of Danish surface flow constructed wetlands.

Sites (n)	Years	Removal rate (kg ha^{-1} y^{-1})		Removal efficiency (%)	
		TN	TP	TN	TP
13	44	472 ± 372	31 ± 26	23 ± 10	45 ± 20

Years: number of years monitored. After Hoffmann et al., 2020.

For the half year periods of July 2017 to June 2020 at Fillerup, the total nitrogen (TN) inlet varied between 206 and 642 kg, while the outlet varied between 193 and 631 kg. Resulting in a TN removal of 13-167 kg, or 44-560 kg/ha, and a removal efficiency of 6.3-26 % (Table 2). The total phosphorus (TP) inlet varied between 1.67-11.3 kg, while the outlet was 0.670-5.98 kg and had a removal efficiency between -1.6 and 60%. The CW at Fensholt received 347-615 kg of TN and 5.0-29.8 kg of TP, while 52-107 kg TN and 2.0-7.8 kg TP were measured at the outlet. Resulting in efficiencies of 13-19% and 36-62% for TN and TP, respectively.

A review by Carstensen et al. 2020 found that CWs treating agricultural drainage water had TN removal means of 41% within a range of -8 to 63% (Carstensen et al., 2020), which is a higher removal mean than for the two CWs of this study. However, studies of Swedish CWs under similar climatic conditions have found removal efficiencies of 3-15 % (Bastviken et al., 2009; Strand & Weisner, 2013). The CWs at Fensholt and Fillerup were both constructed as full-scale testing facilities and knowledge gained have been used to determine the standard for CWs in Denmark. Today CWs must be 1-1.5% of the drained catchment to ensure a minimum HRT of 24 hours in winter, where the largest drainage discharges occur. Fillerup and Fensholt are 0.78 and 0.74% of their catchment areas, respectively, and a larger CW area would improve the efficiencies.

Table 2. Total nitrogen (TN) removal and total phosphorus (TP) retention at the surface flow constructed wetlands of Fillerup and Fensholt, Jutland, Denmark.

Study site	Period	TN removal (kg/ha)	TN- removal (%)	TP retention (kg/ha)	TP retention (%)
Fillerup	07/17-12/17	560	26	17.8	47
	02/19-06/19	44	6.3	3.36	60
	07/19-12/19	289	12	14.8	55
	01/20-06/20	235	16	-1.0	-3.6
Fensholt	07/17-12/17	437	19	31.8	36
	02/19-06/19	212	15	8.16	40
	07/19-12/19	327	13	24.9	43
	01/20-06/20	314	19	53.1	62

The average TP retention efficiencies of Fillerup and Fensholt (Table 2) are higher than the general average for CWs of 33%, ranging from -103 to 68% (Carstensen et al., 2020). The danish CWs generally show high TP retentions with an average efficiency of 45% (Table 1) (Hoffmann et al., 2020). It is noteworthy that soluble reactive phosphorus had a net retention of 67 and 54% for the whole period, while particulate P had a net loss of -66 and -97% for Fillerup and Fensholt, respectively. This contradicts the findings of Mendes et al. (2018).

Further investigations are needed to evaluate the processes controlling P dynamics in the CWs.

CONCLUSIONS

Surface flow constructed wetlands treating agricultural drainage waters has been successfully implemented in Denmark. Knowledge gained at the first CWs have been used to optimize the design and implement CWs as the first Danish mitigation measure targeting local environmental needs.

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Small-scale water reservoirs as hotspots for GHG emissions

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INTRODUCTION

Surface waters are important components of the global carbon cycle (Bastviken et al, 2011). Especially water retention reservoirs are known to be sources of carbon dioxide (CO_2) and methane (CH_4) and thus they are contributing to the greenhouse effect and climate change (Deemer et al, 2016). Where most studies until now focused on large reservoirs, the carbon emissions of smaller reservoirs are often overlooked (Gómez-Gener et al, 2018). Therefore, the aim of this study was to determine the importance of carbon emissions from small-scale reservoirs.

METHODS

The research was done in two reservoirs along the Berkel river in the Netherlands, which were constructed in 2013 for water retention and nature development. The total area of the reservoirs is around 1.7 ha and the depth ranges from 0.3 to 0.8 m, During a one-year period, diffusive GHG emissions from the river and the reservoirs were measured monthly, using a floating chamber connected to a Picarro GasScouter. The measurement locations are given in figure 1. Also, the ebullition of CH_4 was measured three times for a period of two days, by an inversed funnel method (Van Bergen et al, 2019). The GHG fluxes from the reservoirs and the Berkel river were compared to determine whether the construction of the reservoirs lead to enhanced carbon emissions.

Fig. 1. Locations of GHG emission measurements.

RESULTS and DISCUSSION

The annual average emissions from the two reservoirs were 1954 mg CO_2/m^2/day and 282 mg CH_4/m^2/day. Ebullition contributed 96% to the total CH_4 flux. CO_2 emissions where the highest in the river, but ebullition was about 74 times as high in the reservoir as it was in the river. Consequently, the reservoir has a larger impact on the greenhouse effect in terms of CO_2 equivalents compared to the river. The reservoirs emit about 9.0 g CO_2-eq/m^2/day on average versus 3.6 g CO_2-eq/m^2/day from the river. Figure 2 illustrates the relative importance of the three different carbon fluxes.

The CO_2 and CH_4 emissions from the reservoirs were both more than twice as high as the global average emissions from reservoirs estimated by Bastviken et al (2011) and Deemer et al (2016). Extrapolated to all surface waters in the Netherlands this CH_4 emission exceeded the total human caused CH_4 emission in the country.

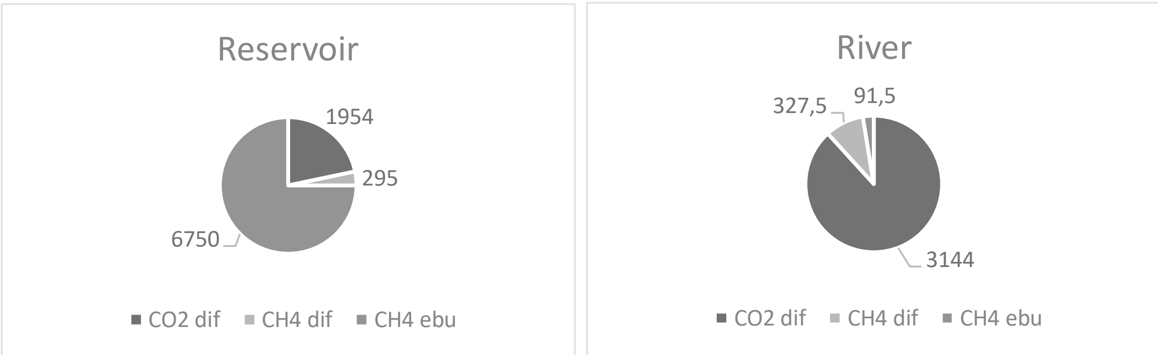

Fig. 2. Average diffuse and ebullitive carbon fluxes from the Berkel reservoirs and from the river itself in terms of mg CO_2 equivalents per m2 per day.

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Fishing for methane: Ebullition is the main pathway of methane emissions from freshwater fish ponds

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INTRODUCTION

Fish are important sources of protein, essential fatty acids and micronutrients in the human diet. Accordingly, the combination of ever-increasing global demand for animal-source foods and dwindling wild fish stocks led to a boom in the aquaculture sector in recent years (FAO, 2020). Due to a high input of organic matter and nutrients by fish feed and/or fertilization, aquaculture ponds are prone to be substantial sources of methane (CH_4) (Rosentreter et al., 2021; Yuan et al., 2019). CH_4 is a greenhouse gas that has a global warming potential that is 34 times higher than carbon dioxide on a 100-year time horizon (Myhre et al., 2013). Very little is known, however, about the magnitude, pathways, and drivers of CH_4 emissions from aquaculture ponds. Particularly the ebullitive (bubble) flux of CH_4 from fishponds is often not quantified, even though it may contribute substantially to total emissions (Kosten et al., 2020).

METHODS

We studied CH_4 emission in 52 fishponds located in 21 freshwater fish farms in Brazil (fig. 1), where aquaculture production has increased over a hundredfold since 1995 (FAO, 2018). We measured diffusive CH_4 fluxes at the margin and the centre of each pond using a closed chamber connected to a greenhouse gas analyser. In a subset of ponds (n=28) we additionally quantified ebullitive CH_4 fluxes using inverted funnels which were deployed for 24 hours. Additionally, we sampled water and sediments for biogeochemical analyses.

Fig. 1. Locations of the 21 fish farms in south-east Brazil (*Google maps, 2020)*.

RESULTS and DISCUSSION

Ebullitive fluxes contributed substantially (median 82%, average 61%) to total CH_4 emissions, surpassing diffusive fluxes in 19 of 28 ponds. Ebullitive emission rates were highly variable, ranging from 0 to 450 mg m^{-2} d^{-1} (median 66 mg m^{-2} d^{-1}, average 116 mg m^{-2} d^{-1}). Extreme fluxes are comparable to rates commonly observed in hypereutrophic aquatic systems. On the other hand, we also observed four ponds without any CH_4 ebullition. Diffusive CH_4 fluxes were relatively low (median 8.4 mg CH_4 m^{-2} d^{-1}) but varied substantially between ponds (0.7 to 85.8 mg CH_4 m^{-2} d^{-1}); Additionally, CH_4 diffusion was higher in the centre (median 11.4 mg CH_4 m^{-2} d^{-1}) than at the margins (median 6.4 mg CH_4 m^{-2} d^{-1}) in 90% of ponds (p<0.001).

CONCLUSIONS

Our study shows that CH_4 ebullition is the main pathway of fishpond CH_4 emissions. This implies that the carbon footprint of farmed fish production is currently underestimated as ebullition of CH_4 is not taken into account in most assessments. The large variation in CH_4 emissions between and within fishponds may be explained by different production practices, sediment quantities, and trophic state of the pond. Unravelling drivers for these emissions is essential to support climate-smart management practices in the fast-growing aquaculture sector.

ACKNOWLEDGEMENTS

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Assessment of Blue Carbon Stocks of North Bull Island, Dublin, Ireland

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INTRODUCTION

Coastal wetlands have been shown to be among the most efficient ecosystems for storing carbon on a per area basis - retaining organic carbon for centuries to millennia (Chmura *et al.*, 2003; Duarte *et al.*, 2005; Fourqurean *et al.*, 2012; Ellison and Beasy, 2018; Alongi, 2020). The carbon stored in these ecosystems has been termed "blue carbon" and blue carbon ecosystems (BCEs) specifically refer to saltmarsh, seagrass and mangrove habitats (Nelleman et al., 2009). These habitats store as much as 71% of the carbon found in ocean sediments though they cover less than 0.5% of the seabed (Nellemann *et al.*, 2009). Ireland has 100km^2 of saltmarsh habitat countrywide with varying substrate types (Curtis and Sheehy Skeffington, 1998; Mcowen *et al.*, 2017). Global analyses of saltmarsh carbon densities do not contain data about Ireland's coastal wetlands (Chmura *et al.*, 2003; Ouyang and Lee, 2014). The aim of this study is to provide, to our knowledge, the first account of the carbon density of an estuarine saltmarsh in Ireland.

METHODS

North Bull Island is located in North Dublin Bay, Ireland (53.367065°N 6.148560°W). The island has formed over the past 200 years and is now an important site for nature conservation.
Soil cores were collected from this site using a gouge auger of 100 cm length and 6 cm diameter to measure the soil and belowground biomass carbon pools, and a quadrat (30x30 cm) of vegetation was removed to measure the aboveground biomass. Soil organic carbon was determined using the Loss on Ignition method outlined in the Coastal Blue Carbon manual (Howard *et al.*, 2014).

RESULTS and DISCUSSION

The estimated total carbon stock of North Bull Island was calculated to be 105 953 ± 729 Mg C_{org} with a carbon density of 888 ± 225 Mg C_{org} ha^{-1}. The carbon density calculated for the soil carbon pool is more than twice the average of saltmarshes globally.

Table 1. The Total Organic Carbon (TOC) ± SD (Mg C_{org}) and carbon density ± SD (Mg C_{org} ha^{-1}) of the saltmarsh's main carbon pools

Carbon Pool	TOC (Mg C_{org})	Carbon Density (Mg C_{org} ha^{-1})
Soil	105 454 ± 729	883 ± 225
Aboveground	478 ± 5	4 ± 2
Belowground	21 ± 0.2	0.18 ± 0.001
Total	105 953 ± 729	888 ± 225

CONCLUSIONS

Through this study we have determined the carbon stocks of the North Bull Island saltmarsh to be higher than the global average. To investigate whether this is common among Irish coastal wetlands, further studies will be conducted as part of a larger project.

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