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ASSESSING THE ROLE OF PIT LAKES IN RIVERINE LANDSCAPE REHABILITATION: A COMPREHENSIVE STUDY IN THE PO RIVER BASIN VIA MULTI-SOURCE SATELLITE AND IN SITU DATA

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Abstract

Since the 1950s, an intense sand and gravel quarrying from the river floodplains led to the formation of pit lakes (PLs) which now dot floodplains and have changed the morphology and the drainage system of several Italian river basins. In the last decade, a renewed interest in PLs has grown as tools for rehabilitating rivers and their floodplains, in order to provide a similar ecological service once offered by pristine riverine wetlands, which were lost due to anthropogenic pressures. Despite the potential importance of these environments, systematic data on their location, morphology, and water quality remains limited. The primary aim of this study was to quantify the number, distribution, and major morphometric characteristics of PLs in the Po River basin, in order to assess their relevance and suitability for ecological restoration purposes. To achieve these aims, a synergistic approach was used including remote sensing imagery (Landsat, Sentinel-2, and PRISMA), satellite archives, and regional databases. In addition, by exploiting the synoptic and multispectral view of these satellites, we evaluated the physical and optical properties of a large sample of PLs from 1990 to 2021. Specifically, we focused on two key parameters of water quality: the dominant wavelength (λ_{dom}, i.e., the water colour) and the concentration of Suspended Particulate Matter (SPM), splitting it into its inorganic (Suspended Particulate Inorganic Matter, SPIM) and organic (Suspended Particulate Organic Matter, SPOM) fractions. The results show that more than 12,650 small lentic water bodies are present in the Po River basin, with agricultural ponds the most represented category, followed by PLs. Specifically, 1580 PLs were identified, divided into three subcategories: active (338), ceased (1188), and doubtful (54). These artificial aquatic systems, when compared with pristine riverine wetlands, are more numerous, widely distributed, and on average more extensive, although more regularly shaped. The temporal analysis of satellite imagery showed that the number and the total area of PLs have increased from 1990 to 2021. PLs currently account for 63.5 km^2 surface area, an average water volume of $378x10^6$ m^3, and for a removal of 26.6 mg N m^{-2} d^{-1} via denitrification. The results also show a very strong correlation between SPM concentrations obtained *in situ* and those obtained from satellite images, both for data derived from Landsat (R^2 = 0.85) and Sentinel-2 images (R^2 = 0.82). In general, it appeared that PLs with the highest mean SPM concentrations and the highest mean λ_{dom} (i.e., poor water quality) are located along the main Po River, and more generally near rivers. Active PLs exhibit a poor water quality status, especially those of small size (<5 ha) and directly connected to a river. Seasonal comparison shows the same trend for both SPM concentration and λ_{dom}: higher values in winter gradually decreasing until spring-summer, then increasing again. In addition, it emerged that the end of quarrying activity led to a reduction in SPM concentration from a minimum of 43% to a maximum of 72%. Finally, we demonstrated that the PRISMA hyperspectral satellite is a reliable tool for estimating the inorganic and organic fraction of SPM concentration. In this context, the use of multi-source satellite imagery enabled the quantification and analysis of PLs and other water bodies in a vast area such as the Po River basin, as well as the evaluation of the temporal evolution of the physical and optical properties of the PLs. In particular, the Sentinel-2 images consistently proved to be a reliable resource for capturing episodic and recurring quarrying events and portraying the ever-changing dynamics of these aquatic systems.

Riassunto

L’obiettivo principale di questo progetto è valutare la rilevanza dei laghi di cava a scala di bacino, la qualità delle loro acque e identificare i principali fattori che concorrono a regolarla, integrando approcci metodologici che fanno riferimento alla limnologia classica, con le potenzialità multispettrali e spazio-temporali delle tecniche di telerilevamento. Lo scopo ultimo è di valutare se e in quali condizioni questi sistemi acquatici artificiali sono in grado di sostituire le funzioni e i servizi comunemente associati agli ecosistemi acquatici naturali di pianura (es: lanche fluviali e zone umide). In particolare, lo studio è stato condotto avendo come riferimento spaziale il bacino idrografico del fiume Po.

La prima parte di questo progetto è incentrata sul censimento di tutte le informazioni reperibili relative agli ambienti acquatici del bacino del Po attraverso sistemi informativi geografici regionali e immagini satellitari a media risoluzione. In particolare, sono state processate dodici immagini Sentinel-2 (S2) Livello 2 (L2, ovvero già corrette atmosfericamente) relative alla fine del 2021. A queste immagini sono stati applicati due indici spettrali allo scopo di individuare tutti i pixel acquatici: RWI (Relative Water Index) e WAVI (Water Adjusted Vegetation Index). Questa analisi ha fornito un primo screening dell’area di interesse e ha permesso di identificare 12,685 corpi idrici suddivisibili in nove macro-categorie in base all’origine, alla posizione e alla destinazione d’uso. Di questi, 1580 sono stati classificati come laghi di cava e 1111 come zone umide e/o lanche fluviali, mentre la categoria più rappresentata risulta essere quella dei cosiddetti “ponds” agricoli (7252). I laghi di cava sono stati a loro volta suddivisi in base all’attività estrattiva e ad oggi sono stati identificati 338 laghi di cava attivi, 1188 laghi di cava cessati e 54 laghi di cava difficilmente classificabili. Per ciascuno di essi è stato calcolato il perimetro, l’area e l’indice di sinuosità. Successivamente è stato scelto un sottocampione eterogeneo di 320 laghi di cava suddivisi in otto aree differenti: Brescia, Mantova, Torino, Parco fluviale del Po e dell’Orba, Asta del Po, Modena, Trezzo sull’Adda e Milano. A seconda della fascia fluviale di competenza, sono state create attorno ai laghi del sottocampione delle zone buffer di 500 m (all’interno della zona golenale del Po) e 250 m (al di fuori di essa). Queste zone buffer sono state utilizzate per valutare il cambiamento dell’uso del suolo dagli anni ‘90 fino al 2018 e per individuare la vegetazione ripariale nei dintorni dei laghi. Per identificare quest’ultima è stato applicato l’indice spettrale SAVI (Soil Adjusted Vegetation Index) alle immagini S2 L2 del 2021 ed è stata condotta una classificazione mediante lo strumento “maximum likelihood” del software ENVI v.5.7. Questa prima parte del progetto ha portato alla stesura di un paper pubblicato sulla rivista “*Ecological Engineering*” intitolato: “*Pit lakes from gravel and sand quarrying in the Po River basin: an opportunity for riverscape rehabilitation and ecosystem services improvement*”. Dai risultati di questo studio è emerso che i laghi di cava, se confrontati con le zone umide, sono più numerosi, ampiamente distribuiti e mediamente più estesi, anche se di forma più regolare. Dall’analisi temporale è emerso che il numero e la superficie totale dei laghi di cava sono aumentati dal 1990 al 2021, raggiungendo una superficie totale di 63.5 km^2 e un volume medio di $378x10^6$ m^3.

La seconda parte di questo progetto è invece dedicata all’analisi delle proprietà fisiche ed ottiche di questi sistemi acquatici artificiali. Nello specifico, è stata valutata l’evoluzione temporale (1990-2021) della concentrazione di SPM (Suspended Particulate Matter) e della lunghezza d’onda dominante (λ_{dom}, ovvero il colore dell’acqua) in un ampio sottocampione di laghi di cava (320, suddivisi nelle otto aree citate in precedenza). Questi due parametri, essendo intrinsecamente legati alle caratteristiche ottiche dell'acqua, sono stati impiegati come indicatori preliminari per valutare lo stato di qualità delle acque dei laghi di cava. Tuttavia, è fondamentale tener presente che, pur essendo molto importanti, non sono esaustivi nel definire la qualità di questi ambienti.

Considerato l'ampio intervallo temporale utilizzato, è stato necessario utilizzare tre satelliti differenti: Landsat-5 (L5), Landsat-7 (L7) e Sentinel-2 (S2). Quando disponibili, sono state scaricate sei immagini per ciascun anno: inverno (1 gennaio - 20 marzo), primavera (1 aprile - 15 maggio), primavera-estate (16 maggio - 15 giugno), estate (1 luglio - 31 agosto), estate-autunno (1 settembre - 1 ottobre) e autunno (2 ottobre - 31 novembre). In totale sono state scaricate e processate 375 immagini satellitari Livello 1 (L1, ovvero non corrette atmosfericamente). La rete neurale ACOLITE v.2022 è stata utilizzata per la correzione atmosferica e per la stima della concentrazione di SPM per ciascun lago di cava. Inoltre, a partire dalle Remote Sensing Reflectances (Rrs) è stata calcolata la λ_{dom} tramite l'indice Forel-Ule. Per validare i prodotti di qualità delle acque sono state effettuate delle campagne di misura *in situ*, sincrone al passaggio del satellite, durante le quali sono stati raccolti campioni d'acqua e sono state acquisite le firme spettrali (Rrs). Una volta processate tutte le immagini satellitari, sono state create delle ROI (Regions of Interest) ad hoc per ciascun lago di cava, grazie alle quali sono state estratte le concentrazione di SPM e le λ_{dom}. Infine, i laghi di cava sono stati divisi in quattro gruppi tematici con lo scopo di comprendere se la concentrazione di SPM e la λ_{dom} siano influenzati dalla posizione del lago, dalle dimensioni, dalla stagione e dall'attività estrattiva. Questa seconda parte del progetto ha portato alla stesura di un paper pubblicato sulla rivista "*Remote Sensing*" intitolato: "*Long-term detection of SPM concentration and water colour in gravel and sand pit lakes through Landsat and Sentinel-2 imagery*". Dai risultati di questo studio è emerso che esiste un'ottima correlazione tra i dati *in situ* e i dati ottenuti dalle immagini satellitari, sia per quanto riguarda la concentrazione di SPM sia per la λ_{dom}. In generale, è stato dimostrato che i laghi di cava con una maggiore concentrazione di SPM e una λ_{dom} media più elevata sono situati lungo il fiume Po. È risultato inoltre evidente come i laghi di cava la cui attività estrattiva non è ancora cessata, presentino uno stato di qualità delle acque scadente, soprattutto quelli di piccole dimensioni (<5 ha) e direttamente collegati ad un fiume. Il confronto stagionale conferma lo stesso andamento sia per la concentrazione di SPM che per la λ_{dom}: valori più elevati in inverno che diminuiscono gradualmente fino alla primavera-estate, per poi aumentare nuovamente. È emerso infine come la cessazione dell'attività estrattiva abbia comportato una riduzione della concentrazione di SPM da un minimo del 43% ad un massimo del 72%.
Infine, ad integrazione di quest'ultimo lavoro è stato testato il satellite iperspettrale PRISMA (PRecursore IperSpettrale della Missione Applicativa) con l'obiettivo di stimare la frazione inorganica (SPIM) ed organica (SPOM) dei solidi sospesi. È stato selezionato un ristretto campione di laghi di cava e la loro componente dominante dei solidi sospesi è stata ipotizzata sulla base dell'attività estrattiva, della presenza di vegetazione ripariale e sulla quantità di campi agricoli nei loro dintorni. Sono state scaricate tutte le immagini PRISMA (sia L1 che L2) disponibili che avessero un'immagine S2 sincrona (discrepanza massima di 2 giorni). Le concentrazioni di SPM sono state stimate dalle immagini S2 tramite la rete neurale ACOLITE, mentre per le immagini PRISMA le concentrazioni di SPM, nonché le percentuali di SPIM e SPOM, sono state stimate tramite il modello bio-ottico BOMBER (Bio-Optical Model-Based tool for Estimating water quality and bottom properties from Remote sensing images). Dal confronto tra le concentrazioni di SPM è emerso un ottimo accordo tra i prodotti S2 e quelli PRISMA, mentre il confronto tra le percentuali di SPIM e SPOM con la componente dominante dei solidi sospesi ipotizzata ha evidenziato che il modello bio-ottico BOMBER è in grado di stimare correttamente le due frazioni nella maggior parte dei laghi di cava (47 su 53 laghi analizzati). Laddove è stata riscontrata una discordanza, le due frazioni erano comunque in equilibrio, con valori prossimi al 50%.
In conclusione, l'impiego di immagini satellitari multi sorgente ha permesso di identificare, quantificare e analizzare i laghi di cava in un'ampia area come il bacino del fiume Po, consentendo inoltre di valutarne l'evoluzione temporale. In particolare, le immagini S2 si sono dimostrate una

risorsa affidabile per catturare eventi estrattivi episodici e ricorrenti, offrendo così una rappresentazione accurata delle dinamiche in costante mutamento tipiche di questi sistemi acquatici artificiali.

Organization of the chapters

Chapter I of this Ph.D. thesis introduces the main topic: pit lakes (PLs) and their relevance at the basin-scale. Initially, I outlined the rationale behind the choice of this topic, emphasizing its relevance in the scientific context, and highlighted the main advantages of using remote sensing techniques to study these artificial aquatic systems. Afterward, I described the main characteristics of these lakes, dwelling on the problems and opportunities associated with them. I then provided a summary of the scientific literature of the last four decades, updated to 2023, concerning gravel and sand PLs. Finally, I presented the side activities and research, which constituted the basis for the design and development of this thesis.

Chapter II focuses on the identification of PLs located in the Po River basin, highlighting their distribution and morphometric characteristics (Ghirardi et al., 2023a). The main aim was to assess the relevance and suitability of these artificial aquatic systems for ecological restoration. To achieve this goal, a synergistic approach was adopted between remote sensing techniques, satellite archives, and information from regional databases.

Chapter III is dedicated to the assessment of physical and optical properties in a large sample of PLs located in the Po River basin between 1990 and 2021 (Ghirardi et al., 2023b). Specifically, we focused on two main parameters: the concentration of SPM (Solid Particulate Matter) and the dominant wavelength (λ_{dom}, i.e., the water colour). To conduct this analysis, we adopted an approach that combines data obtained through remote sensing (Landsat constellation and Sentinel-2) with classical limnological techniques.

Chapter IV represents an extension of the study conducted by Ghirardi et al. (2023b). Specifically, we performed detailed analyses on a restricted sample of PLs, for which chemical, morphological information and surrounding land use were known. To conduct these analyses, we used the hyperspectral satellite PRISMA (PRecursore IperSpettrale della Missione Applicativa), which can provide much more information than a multispectral satellite (e.g., Sentinel-2). In detail, the main aim was to break down the SPM concentration into two distinct components: the inorganic fraction (SPIM, Suspended Inorganic Particulate Matter) and the organic fraction (SPOM, Suspended Organic Particulate Matter).

Finally, the **Chapter V** contains the overall conclusions and main achievements during the three years of the Ph.D. program.

My Ph.D. thesis represents a contribution to the management of the UNESCO MaB Reserve "Po Grande". This project involved two research units, the Institute for Electromagnetic Sensing of the Environment of the National Research Council in Milan (CNR-IREA, coordinated by Dr. Mariano Bresciani) and the University of Parma (coordinated by Prof. Pierluigi Viaroli). This research did not receive any specific grant from funding agencies in the public, commercial, or not-for-profit sectors.

Chapter I: Introduction

1.1 Rationale and relevance of the study

Sand and gravel deposits stand as significant natural resources with a dual purpose: mining for construction material and pumping for drinking water (Peckenham et al., 2009). However, these aggregates represent a non-renewable mineral resource. Consequently, the extraction of these valuable materials often encounters limitations imposed not primarily by the sheer extent of the resource, but rather by factors like urban expansion (Johnson, 2001), nature conservation and groundwater protection legislation (Botta et al., 2009). Reconstruction and unprecedent economic growth in Europe after the Second World War were supported by heavy exploitation of riverine sand and gravel deposits (Mollema and Antonellini, 2016). In a first phase, most of the aggregates were directly withdrawn from the riverbeds causing deep morphological alterations of the riverscape. Incision of the riverbed was the substantial effect, changing reaches from braided to single-channel and causing the disconnection between water flow and floodplains. The impacts of aggregate exploitation were further amplified by changes in land uses and, overall, by river damming and water abstraction, which deeply altered land-to-ocean fluxes of water, sediment, and nutrients (Meybeck and Vörösmarty, 2005). The floodplain was also impaired by the progressive loss of oxbow lakes and of a variety of wetlands which were no longer fed by river flooding, thus diminishing unique habitats and species, and related functions and ecosystem services (Meybeck and Vörösmarty, 2005; Strayer and Dudgeon, 2010; Darwall et al., 2011; Alofs et al., 2014). In the last two decades, these trends were also exacerbated by climate change, e.g., the shift from perennial to intermittent hydrological conditions (Datry et al., 2017).

In the 1980s, once the serious deterioration of river ecosystems was perceived, quarrying activity was moved from riverbeds to floodplains, causing the formation of shallow lakes (pit lakes, PLs) superficially resembling aspects of pristine wetlands and oxbow lakes (W&Os). These lakes might provide an unprecedented opportunity to rehabilitate the riverscape. In this context, a reliable goal is to restore ecosystem processes and functions to improve the provision of ecosystem services rather than the reconstruction of the pristine reference ecosystems (Gerwin et al., 2022). On the other hand, there is little knowledge about the possible risks they can pose, e.g., for groundwater systems and for river eutrophication. However, based on the current aggregate demand of ~$3x10^9$ t y^{-1} in Europe (UEPG, 2017; Morley et al., 2022), it can be assumed that these artificial lakes will be relevant for river management in the next decade, e.g. with potential to address the main goals of the European Biodiversity Strategy. Furthermore, PLs are now present worldwide and they increasingly affect, with both negative and positive impacts, the landscape of anthropized regions and are therefore a permanent legacy of previous economic development (Garnier and Billen, 1994; Peckenham et al., 2009; Blanchette and Lund, 2016; Mollema and Antonellini, 2016).

As such, PLs could represent an opportunity for the environmental rehabilitation and management of river floodplains (Kattner et al., 2000; Weilhartner et al., 2012; Muellegger et al., 2013; Peckenham et al., 2009; Søndergaard et al., 2018). Here we use the term rehabilitation (Gerwin et al, 2022), PLs being new aquatic systems that can restore processes and improve the functionality of the floodplain. Despite their numerical importance, PLs and small lentic water bodies in general have been neglected in the scientific literature (Cheng et al., 2022) and in national and international laws and regulations, e.g., they are not included in the European Water Framework Directive (WFD, 2006/118/EC). Nevertheless, PLs can represent a new addition to aquatic resources that can help counter the global water resources crisis (Brown 2003).

One of the main limitations in considering the potential for PLs to play a role in the riverscape rehabilitation and to provide ecosystem services is the lack of information on the actual number of

PLs, their distribution, their shape and similarity to natural W&Os, and the anthropogenic pressure and threats acting on them. While land-based studies of small lentic water bodies in a given watershed are expensive and time consuming, satellite remote sensing offers reliable tools for large-scale and repeated surveys of aquatic environments (Ghirardi et al., 2020, 2022; Free et al., 2021; Matta et al., 2022). Remote Sensing allows the remote acquisition of detailed information about the qualitative and quantitative characteristics of aquatic surfaces by exploiting the different way in which they interact with electromagnetic energy. The entire process is based on the use of sensors capable of measuring electromagnetic energy, both reflected and emitted. The use of remote sensing techniques has many advantages including a synoptic, multitemporal and multispectral view, and the ability to perform repeated captures over large areas, overcoming many limitations imposed by *in situ* sampling techniques.

Given the lack of information on PLs and the unreliability of regional databases in Italy, we carried out the present study to investigate these artificial aquatic systems at the basin-scale. This study aims to represent a contribution in the broader context of evaluating PLs as substitutes for lost or degraded W&Os with the final purpose of restoring processes and functions in floodplain of the Po River basin. For this reason, we aimed at assessing the number, distribution, surface areas, main morphometric characteristics, and land uses in the PLs surroundings, and their evolution in the last decades. In addition, we evaluated the physical and optical properties of these lakes, focusing on the calculation of the dominant wavelength (λ_{dom}, i.e., the water colour) and the estimation of concentration of Suspended Particulate Matter (SPM); parameters that can contribute to the assessment of water quality. Water colour is a key parameter for the study of inland waters because it is related to the optical properties of water and is directly linked to changes in water constituents. Another key factor for understanding the dynamics of aquatic ecosystems is the concentration of SPM as it can affect the optical properties of water through absorption and scattering of sunlight. This bio-optical parameter consists of a mixture of inorganic and organic substances and the capability to distinguish between these two components is critical to understanding changes in water transparency and quality.

The Po River basin, where this study was carried out, is a heavily exploited watershed which accounts for 40% of Italian Gross Domestic Product (GDP) over less than 25% of the surface of Italy. By contrast, much of the river reaches are included in five UNESCO Man and Biosphere Reserves. Within this context we speculated on the exploitation of PLs as ecosystem service providers in scenarios of drought and water scarcity (water storage and provisioning) and for the regulation of the nitrogen cycle.

1.2 Characteristics of pit lakes

PLs can be defined as man-made reservoirs that are formed as a result of mining operations and are fed mainly by four sources of water: groundwaters (when the quarry intercepts the water table), rivers and streams (when lakes are located in the river floodplain), surface runoff, and rainfall. Excavation can be conducted in two way: dry or wet dredging. In the former case, the quarry does not intercept the water table and consequently does not receive external water inputs except for rainfall and surface runoff. After the conclusion of mining activities, these quarries can either be filled with inert materials or waste, or they can be linked to surface water bodies to serve as reservoirs. In contrast, with the wet dredging the extraction takes place in correspondence with watercourses and unconfined aquifers whose waters gradually fill the cavity. This second approach offers more efficient resource utilization and minimizes land consumption (Muellegger et al., 2013). These artificial aquatic systems are difficult to classify, and consequently to legislate. The reason behind this complexity lies in the fact that each individual lake possesses a distinctive set of biophysical characteristics that profoundly influence limnological processes, such as: water quality, catchment interaction, size, location, morphology, hydrology, water depth, etc. (Fee et al., 1996). Nevertheless, in a broad sense, it is feasible to categorize various types of PLs based on their hydrogeological context (Mollema and Antonellini, 2016):

- On-level PLs: this type of PLs essentially aligns with the water table, resembling an extension of it. Consequently, there is no lateral flow of water within the lake;
- Terminal PLs: groundwater flows into the lake from all sides since the lake is situated below the water table;
- Flow-through PLs: the water table is on two different levels compared to the lake, consequently there is drainage into and out through a lateral flow of groundwater;
- Connected PLs: apart from the aforementioned isolated lakes, there exist PLs directly linked to rivers or located in close proximity to them. In the former scenario, these PLs receive water inputs from both groundwater sources and directly from the river. In the latter case, except in instances of severe flooding, there exists a continuous flow of groundwater into the lake and into the river, creating a dual groundwater flow system connecting the lake and the river.

In current river basin planning, the design of sand and gravel quarry is primarily based on the optimization of the quarrying activity, with mandatory prescription for only hydraulic safety, excavation depth, and, to a lesser extent, lake morphology. Less attention is given to the end use and fate of PLs, and environmental and ecological goals are not yet included in the quarry design, but typically postponed at the end of the mining activity as an environmental restoration activity. A step forward from these procedures was made when the Emilia-Romagna Region (Italy) implemented guidelines for the environmental restoration of sand quarries in the Po River floodplain, including ecological issues in the quarry design (Rossi et al., 2007).
Surface areas are generally a few hectares, nevertheless, due to increasing demand for resources, the extent of these lakes has increased over time as mining technologies have become more sophisticated and able to operate on a larger scale (Blanchette and Lund, 2016). In terms of depth, PLs are generally shallow (2-12 m) (Mollema et al., 2015a), but in some cases they can reach considerable depths (e.g., 40 m) (Mollema et al., 2015b). As actually designed, PLs tend to have a high depth-to-surface ratio with flat bottoms and steep banks, in order to minimize extraction costs while simultaneously maximizing the amount of extracted materials, that result in a relatively narrow littoral zone, often missing a marsh zone (Mollema et al., 2015a; Blanchette and Lund, 2016). This morphology has direct consequences for the quality of the whole lake ecosystem, as the metabolic and filtering activities of the littoral zone are key functions which regulate water quality (Wetzel, 1990; Sollie et al., 2008; Nizzoli et al., 2014). Water quality, as well as its physicochemical parameters, can vary greatly among PLs depending on location, extent, quarrying activity, destination of use, surrounding geology, and catchment interaction (e.g., presence of riparian vegetation, interaction with groundwater, direct or indirect connection with rivers, etc.). As a result, PLs may have different trophic states, may be alkaline or acidic, fresh or saline, and toxic or nontoxic (Geller et al., 2013). In general, most groundwater-fed PLs are classifiable as oligotrophic, at least for the first few years following the end of quarrying (Kattner et al., 2000), while those located in the watershed of rivers usually exhibit a worse quality status (Cobelas et al., 1990). Another peculiar characteristic of PLs concerns the residence time of water. This parameter has been documented to vary from 0.03 to 0.04 years in PLs directly connected to a river (Cross et al., 2014) and from 0.1 to 2 years for PLs isolated and fed by groundwater (Mollema et al., 2015a,b;

Weilhartner et al., 2012). The residence time of water in PLs is not a static value as it can increase with the age of the lake (referring to the end of quarrying). This gradual increase occurs as bank permeability decreases due to clogging. Clogging is a highly spatially and temporally variable process that varies from lake to lake and is due to an accumulation of suspended solids, organic and inorganic particulate matter, the formation of gas bubble and sediment compaction (Baveye et al., 1998, Hoffmann and Gunkel, 2011).

1.3 Pit lakes: problems and opportunity

The concept of sustainability in the context of the mining industry is quite complex. McCullough and Lund (2006) define it as "minimizing the long-term risks of pit lakes, while maximizing both short- and long-term benefits for all stakeholders". Based on this premise, there has been a growing recognition in recent decades of the need to plan the end of quarrying activities. This planning necessarily involves a preliminary strategy that focuses on the final shape, destination of use, and monitoring of the lake once excavation has ceased (Evans et al. 2003). Consequently, a collaboration between ecologists and the mining industry would allow planning for PLs not only to minimize risks, but also to maximize the benefits associated with these artificial aquatic systems (Blanchette and Lund, 2016).

The creation of PLs can impact the landscape and environment in several ways, both positive and negative. In general, these aquatic systems can play some important ecological and socioeconomic roles; however, anthropogenic impact and mismanagement can lead to a number of issues. Possible issues associated with the mismanagement of these aquatic systems include:

- **Water contamination**
  Pesticides and other pollutants from adjacent agricultural lands can contaminate water through surface runoff (Søndergaard et al., 2018). In addition, the creation of PLs may result in an increased risk of mosquito-borne disease outbreaks (Norris 2004; Doupé and Lymbery, 2005; Mollema and Antonellini, 2016).
- **Alteration of groundwater quality**
  In general, land surface changes due to quarrying activities can affect the quantity and quality of groundwater (Welhan 2001; Drew et al., 2002; Stichler et al., 2008). One of the main consequences is the alteration of groundwater recharge pathways and flows (Peckenham et al., 2009). Another effect is the loss of the protection afforded by soil, as the organic layer located above sand and gravel deposits is able to absorb pollutants by acting as a buffer zone (Rutherford et al. 1992; Kalbitz et al. 2000, Weilhartner et al., 2012). Loss of this important layer can lead to degradation of the aquifer as it is exposed to the atmosphere (Langer and Arbogast, 2001; Drew et al., 2002; Couillard et al., 2008; Muellegger et al., 2013), surface runoff (Bach et al., 2001; Nizzoli et al., 2010), contaminants (Pitois et al., 2000; Brookes et al., 2004), and other pathogens. In addition, a loss of fresh water may occur because the evaporation of surface water is higher than the evapotranspiration of the vegetated soil being excavated (Mollema and Antonellini, 2016).
- **Eutrophication**
  One of the major issues related to PLs is eutrophication. Most of these aquatic systems are located in suburban areas characterised by a strong presence of agricultural activities, consequently the continuous input of nutrients via surface runoff can cause eutrophication (Sutton et al., 2011; Billen et al., 2013) and intense algal blooms that can negatively affect the trophic status and consequently the water quality of these lakes (Alvarez-Cobelas et al., 1992; Codd, 2000; Tavernini et al., 2009, Cross et al. 2014). Furthermore, in cases of

eutrophic conditions, phytoplankton growth and NO_3^- assimilation reduces nitrate concentrations in the water column and light penetration depth, consequently limiting benthic primary production and microbial benthic denitrification (Nizzoli et al., 2020).

On the other hand, several studies suggest that the presence of these water bodies in combination with careful planning and management could lead to different benefits:

- **Water supply**
  PLs represent a huge opportunity for water storage and management, both for mining companies, but more importantly for neighbouring communities and the local environment (Mollema et al., 2015b). This is vitally important because in the coming decades, pressures on watersheds will inexorably increase due to climate change (Oude-Essink et al., 2010). Possible uses include irrigation of crops, livestock watering, industrial uses, and public drinking water supplies (Gammons et al., 2009). In addition, if placed in strategic locations, they can act as flood retention areas (Cross et al., 2014).
- **Recreation and tourism**
  In the absence of site-specific safety issues, once quarrying activity has ceased, the lake can be used for public recreational activities such as swimming, boating, scuba diving, fishing, etc. (Yehdegho and Probst, 2001; Emmrich et al., 2014; Zhao et al., 2016; Blanchette and Lund, 2016). Among the many recreational activities offered by PLs, sport fishing is undoubtedly one of the most popular and can often turn out to be an important contributor to an area's tourism economy. In addition, these artificial environments may attract wildlife by offering exceptional opportunities to observe them. For example, waterfowl can often be common users of PLs (Phylips, 1992), and such a setting would provide birdwatchers the opportunity to observe them.
- **Aquaculture**
  The commercial or recreational use of PLs for aquaculture may prove to be an innovative way to enhance the ecological, economic and cultural sustainability of open-pit mining (Otchere et al., 2004). They may indeed be more suitable for aquaculture than natural lakes because the former usually have no surface connection to any other water body, thus eliminating the risk of introduction of non-native fish or other negative consequences of aquaculture (e.g., disease, antibiotics, fertilizers) (Otchere et al., 2004; Naylor et al., 2005). On the other hand, there are several examples where extensive aquaculture in some PLs had negative impacts on water quality due to eutrophication from nutrient overload (Axler et al., 1996, 1998; Gołdyn et al., 2010; Nizzoli et al., 2010).
- **Improve groundwater quality**
  Although these artificial aquatic systems pose a potential risk to groundwater, it has been demonstrated that PLs can instead improve groundwater quality (Muellegger et al., 2013). Specifically, physicochemical and biological processes within the lake and especially bank-operated filtration play a crucial role (Kedziorek et al., 2008; Massmann et al., 2008; Wiese et al., 2011). Oxygen availability in PLs can support oxidative processes, which are missing in confined aquifers. Moreover, the succession of oxic and anoxic phases can generate coupled redox processes, e.g. the coupled nitrification-denitrification process (Nizzoli et al., 2010, 2020).

- **Role in biogeochemical processes**
  PLs could play a critical role in the biogeochemical cycle of different elements, as they represent highly active sites for the transport, transformation, and storage of matter received from the environment (Wetzel and Likens, 1991; Tranvik et al., 2009; Bastviken et al., 2011; McDonald et al., 2013; Marcé et al., 2015; Nõges et al., 2016), especially with increasing age of the lake (Weilhartner et al., 2012). For example, increased organic matter and sustained growth of aquatic biomass allow PLs to act as sinks for atmospheric CO_2 (Blanchette and Lund, 2016). In addition to carbon storage, PLs play an important role in the broader carbonate-bicarbonate balance: uptake and release of CO_2 (Raymond et al., 2015), weathering of silicate minerals (Liu et al., 2010), photosynthetic uptake of dissolved inorganic carbon (DIC) and assimilation of C, Ca, and Si by aquatic organisms (Einsele et al., 2001; Iglesias-Rodriguez et al., 2008), various processes for the production, anaerobic degradation, and partial oxidation of methane (CH_4) (Bastviken et al., 2011; Weilhartner et al., 2012, Blees et al., 2015), and finally the dissolution of carbonate ($CaCO_3$) by CO_3 and H_2O. In recent years, PLs, as well as shallow water bodies and secondary hydrographic networks, have been recognized as possible sinks for reactive nitrogen (Nr) (Seitzinger et al., 2006; Harrison et al., 2009; Finlay et al., 2013; Castaldelli et al., 2015; Cheng and Basu, 2017; Nizzoli et al., 2018, 2020). One of the components of phytoplankton is cyanobacteria, which can uptake N_2 directly from the atmosphere. It has been demonstrated that phytoplankton species richness can be very high and depends mainly on season, water salinity, and river water input during floods (Garnier and Billen, 1994; Rojo and Alvarez-Cobelas, 1994; Arauzo et al., 1996; Tremel, 1996; Chapman et al., 1997; Sayer and Roberts, 2001; Hindak and Hindakova, 2003; Padisák et al., 2003).

- **Conservation issues**
  Under optimal conditions, PLs are capable of supporting a diversified ecological community, especially in terms of habitat restoration and maintenance. Consequently, the creation of such artificial systems has the potential to increase the number and diversity of aquatic environments in a specific floodplain, providing new habitats and promoting overall biodiversity, especially in areas characterised by agricultural activities and urban settlements (Danielopol et al., 2000; Linton and Goulder, 2000; Williams et al., 2010; Santoul et al., 2009; Emmrich et al., 2014; Mollema and Antonellini, 2016). Because the morphological characteristics of such lakes tend to evolve over time, biological diversity will vary with the age of the water body, with a greater presence of taxa in the more mature stages of the lake (Lipsey and Malcolm, 1981). Species diversity in a PL will be greatly influenced by the pre-existing biological diversity present in other water bodies in the surrounding area. In addition, biocenosis within these lakes will be influenced by a number of complex factors, including the method of filling and the initial physical and chemical conditions of the lake (Tadonléké et al., 2000; BHP 2005). In many cases, littoral zones and associated habitats are often limited or even absent due to the steep profile of these lakes (Kalin et al., 2001). The lack of this productive habitat will limit the establishment of aquatic vegetation and, consequently, the presence of micro- and macroinvertebrates, fish, and other wildlife. The establishment of macrophyte communities in PLs will depend on several factors including the geographic location of the lake, stage of development, water quality characteristics, slope morphology and light availability. In PLs, as well as in shallow natural lakes, submerged macrophytes play a key role in maintaining a high standard of water quality (Jeppesen et al., 1999; Cross et al., 2014; Rodrigo et al., 2015). Furthermore, macrophyte communities provide habitat and food for zooplankton and

zoobenthos, thus increasing overall biodiversity (Mollema and Antonellini, 2016; Søndergaard et al., 2018). In terms of the chemical cycle, they can accumulate specific elements such as P, K, and Mn (Dykyjova et al., 1998). They also play a significant role in the Ca cycle (Pokorný and Květ, 2003), and are crucial, along with phytoplankton and periphyton, in determining the oxygen regime in the lake (Mollema and Antonellini, 2016). Once mining has ceased, the macrophyte community may vary, and the age of the lake can affect both the number of species and the overall coverage (Lambert-Servien et al., 2006; Søndergaard et al., 2018). In addition, the spread of these aquatic plants could also be hindered by substrate type (Li et al., 2012) and thermal stratification that may originate in deeper ceased PLs (Azzella et al., 2014). The presence of invertebrates is directly related to the existence of littoral habitat characterised by aquatic vegetation. In fact, if productive littoral zones are established, invertebrates may be quite common. However, in situations where meromixis and anoxic bottom water conditions persist, especially considering the steep conformation of PLs, it is possible for a benthic community to be completely absent (BHP, 2005). Regarding the fish community within PLs, the presence of fish can occur in two main ways: (1) through permanent or occasional connections with other water bodies, or (2) through introduction by humans, who may deliberately increase the biomass and species diversity through activities such as fishing and stocking (Radomski and Goeman 1995). In either situation, if conditions are favourable, fish can be abundant in PLs. A study conducted by Zhao et al., (2016) showed that the composition of the fish community in PLs follows a definite trend related to the aging of the lakes itself: in younger lakes, it is common to find no fish species or only one, while later there is the establishment of some pioneer species that are gradually replaced by species more typical of sport fishing activities. Fish require a wide range of habitats to complete their life cycle, which varies by species, season, life stage and sex. These habitats include areas such as macrophyte meadows, gravel beds, rocky shoals, and structures such as boulders and woody debris accumulations. Compared to natural lakes, PLs, at least initially, show a minimal habitat structural complexity and fish species diversity. Consequently, it is essential that restoration and enhancement of these habitats be an integral part of the initial reclamation process during mine decommissioning, especially if sustainable fisheries are to be promoted within these lakes. Finally, PLs are considered an ideal hotspot for waterfowl during post-breeding and wintering periods.

1.4 State of the art

In this section, I will briefly describe the main works published in the field of PLs study. Specifically, I will focus exclusively on gravel and sand PLs, excluding all so-called acid PLs (used for the extraction of metals), because they are not present in my study area, i.e., the Po River basin. Most of these studies focus on a restricted sample of lakes and are often limited both spatially and temporally. Specifically, none of these studies implement remote sensing techniques, as a result, my Ph.D. project appears to be a novel in the context of spatio-temporal analysis of PLs at basin-scale. However, the topics addressed in these works include: eutrophication, management and regulation of these artificial aquatic systems, hydrochemical dynamics, impacts on groundwater, and their role as habitat for a variety of animal (zooplankton, fish and birds) and vegetation (phytoplankton and macrophytes) communities.

One of the first studies dedicated to the hydrochemical analysis of a PL dates back to 1978, when the impact on groundwater of a lake located in Germany was examined, considering both chemical and biological effects (Banoub, 1978). Cobelas et al., (1990) studied the effects of eutrophication on a hypertrophic PL located near Madrid (Spain), observing dissolved oxygen oversaturation, low ammonia, and significantly high nitrate and nitrite concentrations, probably due to the high nitrification rates. A few years later Helmer and Labroue (1993) estimated the denitrification process in several PLs located in France and evaluated their potential for nitrogen dissipation, showing that the main factors influencing denitrification are substrate supply (nitrate and carbon availability), oxygen presence, and temperature. With the aim of understanding if PLs had an impact on groundwater quality, Weilhartner et al., (2012) evaluated the mass balance of nitrate and phosphate as well as the whole ecosystem metabolism in five PLs located in Austria. The results indicated that PLs have the potential to significantly reduce nitrate and phosphate concentrations, and consequently contribute to the improvement of groundwater quality, especially in areas dominated by agricultural field that exponentially increase concentrations of these nutrients. A year later, a similar study was conducted on the same Austrian PLs (Muellegger et al., 2013), which confirmed that the waters of these lakes host highly active biological systems in which primary producers played a crucial role in significantly reducing nitrate inflow concentrations. In this regard, another study confirmed the potential of PLs in contributing to groundwater denitrification (Mollema and Antonellini, 2016). However, the water budget demonstrated that PLs cause freshwater loss in temperate and Mediterranean climates. Søndergaard et al., (2018) evaluated the biochemical status of several PLs located in Denmark and compared them with natural Danish lakes of similar size. In general, PLs appeared cleaner and with lower nutrient concentrations than their natural counterparts. In addition, the littoral zones of all PLs were characterised by abundant populations of submerged macrophytes, with particularly high coverage in the shallow ones. However, most PLs were deeper and with steeper banks than the natural ones, resulting in a limit to the area potentially covered by macrophytes. Finally, Nizzoli et al., (2020) conducted a study to assess the main nitrogen pathways in five PLs of different trophic status located along the Po River (northern Italy). Benthic nitrogen fluxes and denitrification rates were determined in the hypolimnion, and denitrification and reactive nitrogen assimilation by microphytobenthos in the littoral zone. The results showed that the sedimentary littoral zone was the area with the highest total nitrogen removal, although it generally represents a small fraction compared to the total surface area of PLs. In particular, nitrogen removal rates were among the highest reported in the literature.

These artificial aquatic systems can provide ideal habitat for a variety of micro and macro-organisms. One of the first study dedicated to the analysis of plankton communities is that of Backhaus and Banoub (1988) in which a PL located in Germany was investigated. In a similar study, phytoplanktonic communities were analysed in thirteen PLs located near Madrid (Spain), during spring mixing and summer stratification phases (Arauzo et al., 1996). The results showed that during spring, abundance of soluble reactive phosphorus and the presence of significant thermal stability promoted greater biomass development; whereas during summer, excessively high temperatures had negative effects on communities in warmer lakes. Tavernini et al., (2009) conducted two closely related studies in two PLs located in the Po River basin (Italy). The two lakes studied, Ca' Morta and Ca' Stanga, shared a common origin and are contiguous, but they have different ages, and in the former, quarrying activity was still active. In the first study, the seasonal and interannual dynamics of phytoplankton over three years were analysed. No significant differences in species richness were observed between the lakes, but a reduction in biodiversity was observed in Lake Ca' Morta after the excavation phase. In the second study the seasonal and interannual dynamics of zooplankton in the same two lakes were investigated. Significant differences were found in the abundance of zooplankton and the microcrustaceans. In

particular, the lower densities of *Cladocerae* and the presence of large species in the lake still being dredged appeared to be related to the resuspension of sand in the water column. In general, dredging activity, together with some physicochemical parameters of the water, represent the main factors influencing the composition of phytoplankton and zooplankton communities. On the other hand, regarding macro-organisms, Santoul et al., (2004) studied the avian community in six PLs located in the Garonne floodplain (France), while Zhao et al., (2016) quantified taxonomic diversity and fish community composition in seventeen PLs located in southwestern France. The results showed that the presence of submerged macrophytes was the most significant factor in positive terms for the avian community. In contrast, the presence of anthropogenic disturbances had a negative impact on both the total number of birds present and the diversity of species observed; while the fish community composition followed a predictable shift along environmental gradients associated with the age of PLs and the human practice.
Finally, there are studies dedicated to the planning and management of these artificial aquatic systems, both during the excavation phase and after its conclusion. For example, McCullough and Lund (2006) attempted to highlight how planned management of PLs and their end use can benefit businesses, communities, and the environment. Specifically, this study focused on the possible environmental and social risks of PLs (if inappropriately managed), but more importantly on the possible opportunities they can offer once quarrying activity is over. On the other hand, Blanchette and Lund (2016) tried to understand what might be the best destination uses and methods for rehabilitating PLs once quarrying is over. In this regard, the authors argue that PLs pose a unique challenge for environmental sustainability because (1) there is often no clear vision on the end use of these lakes, (2) during planning and design of PLs, ecologists are often not involved, increasing the complexity of rehabilitation and limiting the provision of the ecosystem services, and (3) PL ecology struggling to reach the primary literature. For these reasons, they suggest how PLs should be designed to emulate the key components of a natural lake as much as possible. In particular, the formation of appropriate littoral areas and integrated catchments should be considered.

1.5 Side Research and activities

In this section, I will provide a brief chronological overview of the studies I published as first author or co-author, which were relevant to the drafting of this Ph.D. thesis. In particular, I will focus on the remote sensing techniques employed, which were also applied in this research:

- **Ghirardi et al., (2020)**
  This study focuses on the use of satellite remote sensing to map coastal erosion vulnerability in two Italian sites: Pianosa Island (Tuscany) and Piscinas (Sardinia). We used the first seven bands of the VIS-NIR region (VISible-Near InfraRed) of Sentinel-2 (S2), all reprocessed at the spatial resolution of 10 m. Before processing the optical images, a comparison between Level 1 (L1, not atmospherically corrected) and Level 2 (L2, atmospherically corrected with Sen2Cor) products was performed. L1 images were atmospherically corrected using 6SV code (Second Simulation of the Satellite Signal in the Solar Spectrum code-vector version) (Vermote et al., 1997, 2006). The 6SV output was chosen as reference due to its good performances in retrieving water leaving reflectance in inland and coastal waters (Giardino et al., 2015; Bresciani et al., 2018). The Rrs (Remote Sensing Reflectances) products were then processed using the Sen2Coral add-on-tool (Pianosa) and the non-linear optimization algorithm called BOMBER (Giardino et al., 2012) (Piscinas) in order to obtain maps of three different substrates (sand, rocks and phanerogams) and bathymetry.

Through this study, I gained the expertise to process and use S2 images (both L1 and L2). Specifically, in my study, I used both levels of S2 to identify water bodies and riparian vegetation, and to estimate the SPM concentration and water colour of PLs.

- **Giardino et al., (2020)**
  This study presents a first assessment of the Top-Of-Atmosphere (TOA) radiances measured in the VIS-NIR regions from PRISMA (PRecursore IperSpettrale della Missione Applicativa), the new hyperspectral satellite sensor of the Italian Space Agency (ASI) in orbit since March 2019. In particular, the radiometrically calibrated PRISMA L1 TOA radiances were compared to the TOA radiances simulated with a radiative transfer code, starting from *in situ* measurements of water reflectance (AERONET-OC, WISP Station, PANTHYR). The *in situ* sites represent a rather wide range of water types and trophic conditions but also atmospheric turbidity. Recognising the role of S2 for inland and coastal waters applications, the TOA radiances measured from concurrent S2 observations were added to the comparison. The PRISMA L1 data products were downloaded and re-projected with a geographic lookup table (GLT) for removing artefacts associated with bowtie effects or missing data. The first 57 spectral bands in the 400-900 nm range (spectral bandwidth within ~ 9-12 nm) were extracted to perform the study. Corresponding S2 L1 images were downloaded for a direct comparison with PRISMA. S2 images were processed to transform the dimensionless apparent reflectance into physical units of TOA radiances and for spatially resampling the multispectral cube to 30 m pixel size (comparable to the spatial resolution of PRISMA). The 6SV (Vermote et al., 1997, 2006; Kotchenova et la., 2006) was used in this study to simulate the satellite signal starting from *in situ* measurements of water reflectance. Briefly, the TOA radiances to be compared with PRISMA L1 data were simulated using *in situ* reflectance. The comparison was carried out both as image-by-image, to provide an evaluation for every single scene (across all bands) and band-by-band, to evaluate a set of specific bands common to both PRISMA and AERONET-OC (across all scenes). The results overall demonstrated that PRISMA VIS-NIR sensor is providing TOA radiances with the same magnitude and shape of those *in situ* simulated, with slightly larger differences at shorter wavelengths. The PRISMA TOA radiances were also found very similar to S2 data and encourage a synergic use of both sensors for aquatic applications.

  In this study I had the opportunity to use PRISMA hyperspectral images which, in perspective, can provide much more information than multispectral sensors. Specifically, in my study, PRISMA was used to estimate the inorganic and organic fractions of the SPM concentration of a subsample of PLs.

- **Luciani et al., (2021)**
  This study presents the satellite monitoring system implemented for the SIMILE project and the results obtained in the first 18 months of activities. SIMILE Project has the aim of supporting the definition of management policies and the operational decisions for the insubric Lakes through an innovative information system based on data from an integrated monitoring system, consisting of remote sensing of water quality parameters of the lakes with satellite images, buoys equipped with sensors for the continuous detection of limnological and meteorological parameters within the lakes and in the use of additional data on water quality that will be collected through citizen science. Maggiore, Lugano and Como are the lakes involved in the project. Satellite images acquired by Sentinel-3 (S3), were chosen to produce high frequency maps of water quality products (Chl-a and TSM

concentrations). In addition, when anomalies or unexpected concentrations are detected, a more detailed spatial information is gathered using the higher spatial resolution of the S2 satellites. Finally, the images acquired by Landsat-8 satellite (L8, TIRS sensor) complement the dataset for monitoring the temporal evolution of the lakes surface temperature. The processing of the satellite images to obtain maps of Chl-a and TSM was carried out with ESA's SentiNel Application Platform (SNAP) (Zuhlke et al., 2015). In particular, the radiometric and atmospheric correction of the images and the retrieval of the water quality parameters, was performed with the Case 2 Regional Coast Colour (C2RCC) (Brockmann et al., 2016), an open source processor based on a neural network, implemented as a plugin in SNAP. To check the accuracy of the atmospheric correction performed by C2RCC, the atmospherically corrected reflectances were compared with available *in situ* data, measured between August 2017 and July 2019 in the lakes with a WISP-3 spectroradiometer in the range of 400-800 nm, according to the Seawifs protocol.

In this study, I investigated processing techniques for S2, S3, and L8 images through the use of SNAP software. In addition, I had the opportunity to carry out several measurement campaigns during which I utilized the handheld spectrophotometer WISP-3. In my study, I used SNAP to resample, georeference, and crop satellite images. Furthermore, I used WISP-3 to collect in situ spectral signatures of PLs.

- **Ghirardi et al., (2021)**
  This study investigates the effect of the Adige-Garda spillway opening on the 03/03/2020 on Lake Garda using numerical modelling and maps of Suspended Particulate Matter (SPM) concentration. SPM concentration is one of the many physical characteristics of water bodies that can be assessed by remote sensing, and, in lakes, it is often related to the inflow of riverine waters. Indeed, river inflows carry significant sediment load in lakes and affect water quality not only directly by introducing nutrients, contaminants and thermal energy, but also indirectly by altering the lake's circulation and mixing dynamics. S2 images were used to assess SPM concentrations in the lake before and after the opening event of the 03/03/2020. The atmospheric correction of S2 images was performed using the 6SV code (Vermote et al., 1997, 2006). The 6SV-derived reflectance was converted into remote sensing reflectances (Rrs), which were then used as input for the bio-optical model BOMBER (Giardino et al., 2012), to estimate the SPM concentrations. The thermo-hydrodynamics of Lake Garda were simulated with the hydrodynamic model Delft3D (Lesser et al., 2004) fed by the results of the Weather Research and Forecasting (WRF) atmospheric model (Skamarock et al., 2008). Maps indicate a significant increase in SPM concentrations, especially in the northern part of the lake close to the hydraulic tunnel outlet. Results are consistent with the modelled flow field.

  In this study, I generated several SPM concentration maps from Sentinel-2 imagery; a skill that proved precious in the context of my Ph.D., during which I analyzed the temporal evolution of SPM concentrations in several PLs.

- **Pinardi et al., (2021)**
  In this study, high frequency *in situ* measurements and multispectral satellite data were used in a synergistic way to explore temporal (diurnal and seasonal) dynamics and spatial distribution of Chlorophyll-a (Chl-a) concentration, together with physico-chemical water parameters in a shallow fluvial-lake system (Mantua Lakes). Chl-a is a photosynthetic pigment used to indicate phytoplankton biomass and is considered a proxy to determine the trophic status of lakes (Falkowski and Kiefer, 1985; Steinman et al., 2006; Matthews,

2017). Chl-a concentrations were retrieved from S2 L2 images. In the period between April 2018 and June 2020, a total of 27 images were downloaded for the following processing. The BOA (Bottom-Of-Atmosphere) reflectance was corrected for specular effect and converted in Rrs according to Mobley (1999). Then, the validated algorithm to retrieve Chl-a concentration for the Mantua lakes system parametrized with numerous *in situ* measurements acquired in Mantua Lakes (Bresciani et al., 2013; Bresciani et al., 2017) was applied to all water pixels of the 27 images. The trophic status of the Mantua Lakes system was estimated seasonally and annually using the mean Chl-a concentration in accordance with the OECD classification. The Chl-a concentration maps allowed a seasonal classification of the Mantua lakes system as eutrophic or hypertrophic.

Through this study, I had the opportunity to generate many Chl-a concentration maps from satellite images and to expand my knowledge regarding the assessment of the trophic status of inland aquatic environments.

- **Free et al., (2021)**
  In this study a Non-Parametric Multiplicative Regression (NPMR) was performed to visualize and understand the changes that have occurred between 2003-2018 in Lakes Garda, Como, Iseo, and Maggiore. Specifically, a long series of satellite images were used to obtain the concentration of Chl-a in these lakes. The satellite measurements of Chl-a concentrations used in this study were obtained from four optical sensors that allowed us to cover the temporal range 2003-2011 and 2014-2018. The older set of data were provided by MERIS (Medium Resolution Imaging Spectrometer) onboard of the Envisat-1 platform. The more recent dataset was built with images acquired by optical sensors OLI (Operational Land Imager), MSI (MultiSpectral Instrument), and OLCI (Ocean and Land Colour Instrument), onboard L8, S2, and S3, respectively. For each sensor, satellite observation of Chl-a data were obtained by processing cloud-free at-satellite-radiance data corrected for radiometric noise (e.g., atmospheric effects) to allow the computing of Chl-a from atmospherically corrected images. In the case of newer OLI, MSI, and OLCI imagery the radiative transfer 6SV code (Vermote et al., 1997, 2006) was used to first correct imagery data from atmospheric effects. Chl-a was then retrieved using the tool BOMBER (Giardino et al., 2012), which implemented a bio-optical model that was parametrized with the specific inherent optical properties of subalpine lakes (Giardino et al., 2014; Manzo et al., 2015; Bresciani et al., 2018). Variables that were significant in NPMR included time, air temperature, total phosphorus, winter temperature, and winter values for the North Atlantic Oscillation. Future trends will depend on climate change and inter-decadal climate drivers.

  Also in this study, I had the opportunity to deepen my skills regarding processing techniques for S2, S3 and L8 images. In addition, I experimented with other methodologies for generating Chl-a concentration maps.

- **Gaglio et al., (2022)**
  This study investigated the extent of aquatic emergent vegetation loss for the period 1985-2018 and the consequent effects on carbon sequestration and storage capacity of Valle Santa wetland, a protected freshwater wetland dominated by *Phragmites australis* located in the Po river delta Park (Northern Italy), as a function of primary productivity and biomass decomposition, assessed by means of satellite images and experimental measures. Wetlands are important environmental components for human well-being and sustainable development since provide a multiplicity of ecosystem services and support aquatic

biodiversity (Mitsch et al., 2015; Xu et al., 2020). Nonetheless, they are affected by several pressures and impacts deriving from human activities and climate change (Kingsford et al., 2016; Xi et al., 2021). Particularly, aquatic vegetation loss represents one the most significant forms of ecological deterioration in fresh and brackish wetlands, which can be observed in terms of both reduction of vegetated area and net primary production (i.e., decreased biomass production per spatial unit) (Fogli et al., 2002; Lastrucci et al., 2017; Zhang et al., 2017). Aboveground biomass loss was estimated by the processing and comparing satellite images at different dates after calibration and validation routines based on field measures. The years considered were: 1985, 1989, 1997, 2010, 2016, 2017 and 2018. The images were selected from Landsat and S2 database. After that, they were radiometrically calibrated and converted to surface reflectance after atmospheric correction performed with the 6SV code (Vermote et al., 1997, 2006). Based on research literature of different vegetation index applied to multispectral optical satellite data, six vegetation indexes were used for mapping wetland vegetation. The results showed an extended loss of aquatic vegetated habitats during the considered period. Nonetheless, despite the protection efforts over time, the vegetation loss occurred during the last decades significantly decreased carbon sequestration and storage by 52%, when comparing 1989 and 2018.

In this study, I applied various vegetation indices to satellite images for the identification of wetland helophytes within the Po River basin. In my study, I used different spectral indices to distinguish water pixels and to identify riparian vegetation around PLs in the same study area.

- **Ghirardi et al., (2022)**
  The main aim of this study is to document the variation in spatial extent and density of macrophytes seasonally between 2015 and 2020 of the Sirmione Peninsula (Lake Garda, Italy), using S2 imagery. In addition, our results were compared to previous data from imaging spectrometry; individual parameters affecting macrophyte communities were tested, and the possible effect of the COVID-19 lockdown on macrophyte colonization was evaluated. Macrophytes are crucially important for the functioning of lake ecosystems. They provide structure, habitat, and a food source and are a required component in monitoring programs of lake ecological quality. A total of 34 S2 images available between 2015 and 2020 were processed. In detail, for each year except 2015 (the S2 mission began in July 2015), six images were chosen: two for each season characterised by macrophyte presence (from spring to autumn). The satellite images were converted to TOA radiance and then atmospherically corrected using the 6SV code (Vermote et al., 1997, 2006). Rrs products were then used as input for the BOMBER bio-optical model (Giardino et al., 2012) for estimating the substrate coverage. The BOMBER outputs provide the fractional cover of bottom substrates through two classes: bare sediment and rooted macrophytes. The maps related to each season were merged in average maps representative of the season, as suggested in Roelfsema et al. (2018). Substantial changes were found in both spatial extent and density over the observed period. Variables found to influence the amount of macrophytes included transparency, Chl-a, water level, winter wave height, and grazing by herbivores. A separate analysis focusing on areas associated with boat transit found a recovery in macrophyte coverage during the period of COVID-19 lockdown.

In this study, in addition to processing several S2 satellite images, I investigated the use of the BOMBER bio-optical model for estimating bottom cover and more generally for identifying submerged rooted macrophytes. In my study, I used The BOMBER bio-optical model to distinguish the organic and inorganic fraction of suspended solids.

- **Giacomelli et al., (2023)**
  In this work the wet surface and paths of two stretches of the Taro and Trebbia Apennine rivers were mapped, analysing satellite images from periods with contrasting discharge. The produced images offer multiple possibilities to extract qualitative and quantitative information at the whole stretch scales, including habitat reduction along with decreasing discharge, threshold discharge limiting lateral interactions, or the evaluation of longitudinal river continuity. 21 satellite images (S2 L1) were collected in river stretches within Nature 2000 sites (Directive 92/43/CEE) and processed in order to produce maps of the watercourses under various hydrological conditions, including hydrological extremes. Infrared (IR) spectral bands from the imagery were combined to obtain false color composite images (FCC) useful for highlighting the water occurrence on the ground surface and to distinguish vegetation from bare soil. In addition, bathymetric values were extracted from the same S2 dataset for the two extremes of the spring-autumn time span (i.e., March/April and September/October). According to a multitemporal approach, to quantify the possible changes occurring in the landscape highlighted by the FCC images comparison over the considered time interval, a supervised classification was carried out on each image using the ENVI (Environment for Visualizing Image) software. The image classification was based on the "Maximum Likelihood" algorithm that considers four regions of interest (ROI) identified in each satellite image and corresponding to water, bare soil, vegetation, and soil-and-vegetation. To evaluate changes that could have affected the landscape, the percentage distribution of the four classes was calculated for each image and the values were compared over the time interval considered.

  In this study, I delved into the study of small aquatic environments such as the Taro and Trebbia rivers, with a focus on identifying water pixels and calculating bathymetry. In addition, I used a land use classification tool that I then also employed in the present study.

- **Fabbretto et al., (2023)**
  The aim of this study is to present a broad overview of the main results obtained through the use of hyperspectral reflectance data provided by PRISMA imagery for mapping aquatic ecosystems. Water quality parameters were generated using three different approaches: the bio-optical model BOMBER (Giardino et al., 2012), adaptive semi-empirical algorithms (Nechad et al., 2010; Hestir et al., 2015), and machine learning models (O'Shea et al., 2021; Begliomini, 2022). These methods were tested in a variety of water environments around the world: Curonian Lagoon, Venice Lagoon, Lake Garda, Lake Hume, Lake Mulargia, Lake Peipsi, Lake Trasimeno, and River Promissão. The water quality products (chlorophyll-a, total suspended matter, phycocyanin, and bottom substrate) were then validate with reference data by calculating the Root Mean Square Error (RMSE) and the coefficient of determination (R^2). In general, the results are encouraging for the bio-optical model and the semi-empirical adaptive algorithms, while the products obtained with the machine learning models need further analysis due to the slight statistical agreement obtained.

  In this study, I processed PRISMA imagery with the aim of extracting different water quality parameters. In my study, I applied the BOMBER bio-optical model to PRISMA imagery to estimate the inorganic and organic fractions of the suspended solids in a subsample of PLs.

1.6 Scientific approval

The results of this study were presented at the following conferences:

- XVII incontro dei dottorandi e dei giovani ricercatori in ecologia e scienze dei sistemi acquatici (Napoli, 13-15th April, 2021) – "Assessment of pit lakes in the Po River basin: changes in water quality from satellite images and limnological data". [*Oral presentation*]
- ASITA (2021) – "Identification and characterization of pit lakes at catchment scale using satellite imagery". [*Poster*]
- XXX Congresso della Società Italiana di Ecologia (SItE) (Online 25-27th October, 2021) – "Assessment of pit lakes in the Po River basin: changes in water quality from satellite images and limnological data". [*Oral presentation*]
- Ph.D. School Seminars in Evolutionary Genomics (7-8th July, 2022) – "Assessment of pit lakes at basin scale: changes in water quality from satellite imagery and limnological data in the Po River basin". [*Oral presentation*]
- XXXI Congresso della Società Italiana di Ecologia (SItE) (Siena 13-15th September, 2022) – "Pit lakes from gravel and sand quarrying in the Po River basin: extent, status and ecosystem services potential". [*Poster*]
- XIX incontro dei dottorandi e dei giovani ricercatori in ecologia e scienze dei sistemi acquatici (Online, 12-14th April, 2023) – "Pit lakes from gravel and sand quarrying in the Po River basin: an opportunity for riverscape restoration and ecosystem services improvement". [*Oral presentation*]
- 6th Sentinel-2 Validation Team Meeting (Frascati, 12-14th September, 2023) – "Sentinel-2 application to evaluate the water quality status of pit lakes in the Po River basin". [*Poster*]

Publications on the dissertation topic and side research (role covered in the publication indicated in square brackets; in bold if first author or corresponding author):

- **Ghirardi, N.**, Bresciani, M., De Carolis, G., De Santi, F., Fornaro, G., Giardino, C., ... Zamparelli, V. (**2020**). Mapping of the risk of coastal erosion for two case studies: Pianosa island (Tuscany) and Piscinas (Sardinia). In *Eighth International Symposium "Monitoring of Mediterranean Coastal Areas. Problems and Measurement Techniques"*. 126, 713-722. Firenze. https://doi.org/10.36253/978-88-5518-147-1.71. [*methodology, validation, formal analysis, investigation, data curation, writing—original draft preparation, writing—review and editing, visualization*]
- Giardino, C., Bresciani, M., Braga, F., Fabbretto, A., Ghirardi, N., Pepe, M., ... Brando, V.E. (**2020**). First evaluation of PRISMA level 1 data for water applications. *Sensors*. 20(16), 4553. https://doi.org/10.3390/s20164553. [*methodology, data curation*]
- Luciani, G., Bresciani, M., Biraghi, C.A., Ghirardi, N., Carrion, D., Rogora, M., Brovelli, M. A. (**2021**). Satellite Monitoring system of Subalpine lakes with open source software: The case of SIMILE Project. *BJMC*. 9(1), 135-144. https://dx.doi.org/10.22364/BJMC.2021.9.1.08. [*data curation, validation*]
- **Ghirardi, N.**, Amadori, M., Free, G., Giovannini, L., Toffolon, M., Giardino, C., Bresciani, M. (**2021**). Using remote sensing and numerical modelling to quantify a turbidity discharge event in Lake Garda. *J. Limnol*. 80(1), 1-6. https://dx.doi.org/10.4081/jlimnol.2020.1981.

[*conceptualization, methodology, validation, formal analysis, investigation, data curation, writing—original draft preparation, writing–review and editing, visualization*]

- Pinardi, M., Free, G., Lotto, B., Ghirardi, N., Bartoli, M., Bresciani, M. (**2021**). Exploiting high frequency monitoring and satellite imagery for assessing chlorophyll-a dynamics in a shallow eutrophic lake. *J. Limnol*. 80(3), https://doi.org/1-16. 10.4081/jlimnol.2021.2033. [*data curation, validation, writing–review and editing*]
- Free, G., Bresciani, M., Pinardi, M., Ghirardi, N., Luciani, G., Caroni, R., Giardino, C. (**2021**). Detecting climate driven changes in chlorophyll-a in deep subalpine lakes using long term satellite data. *Water*. 13(6), 866. https://doi.org/10.3390/w13060866. [*data curation, validation, writing–review and editing*]
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- **Ghirardi, N.**, Bresciani, M., Free, G., Pinardi, M., Bolpagni, R., Giardino, C. (**2022**). Evaluation of macrophyte community dynamics (2015-2020) in southern Lake Garda (Italy) from Sentinel-2 data. *Appl. Sci*. 12(5), 2693. https://doi.org/10.3390/app12052693. [*conceptualization, methodology, validation, formal analysis, writing–original draft preparation, writing–review and editing, visualization*]
- Giacomelli, S., Chelli, A., Bresciani, M., Ghirardi, N., Bartoli, M., Marzano, F.N. (**2023**). Mapping Apennines river paths along different hydrological conditions from satellite images: A description of the method and potential applications. *River Res. Appl*. 1-14. https://doi.org/10.1002/rra.4182. [*methodology, data curation, validation, writing–review and editing*]
- **Ghirardi, N.**, Bresciani, M., Pinardi, M., Nizzoli, D., Viaroli, P. (**2023a**). Pit lakes from gravel and sand quarrying in the Po River basin: An opportunity for riverscape rehabilitation and ecosystem services improvement. *Ecol. Eng*. 196, 107103. https://doi.org/10.1016/j.ecoleng.2023.107103. [*methodology, validation, formal analysis, writing–original draft, visualization*]
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- **Ghirardi, N.**, Pinardi, M., Nizzoli, D., Viaroli, P., Bresciani, M. (**2023b**). The Long-Term Detection of Suspended Particulate Matter Concentration and Water Colour in Gravel and Sand Pit Lakes through Landsat and Sentinel-2 Imagery. *Remote Sens.* 15(23), 5564. https://doi.org/10.3390/rs15235564. [*methodology, validation, formal analysis, investigation, data curation, writing–original draft preparation, writing–review and editing, visualization, project administration*]

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Chapter II: Pit lakes from gravel and sand quarrying in the Po River basin: an opportunity for riverscape rehabilitation and ecosystem services improvement

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Pit lakes from gravel and sand quarrying in the Po River basin: An opportunity for riverscape rehabilitation and ecosystem services improvement

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2.1 Introduction and aim of the study

Since the 1950s, rivers and floodplains have been subjected to intensive sand and gravel quarrying. Initially, most of the aggregates were directly withdrawn from riverbeds. Then, once the serious deterioration of river ecosystems was perceived, a large part of the quarrying activity was moved to floodplains during the 1980s, causing the formation of a huge number of pit lakes (PLs). At the same time, due to riverbed incision, floodplains experienced a severe and progressive loss of wetlands and river oxbows (W&Os). In this context, PLs could represent an exceptional opportunity to rehabilitate the riverscape, in fact, once quarrying activities are concluded, PLs may evolve towards aquatic systems whose structure and processes depend primarily on morphology, water depth, connectivity with the river, and land uses in the surroundings. For this reason, they could be exploited as viable substitutes of natural lentic water ecosystems, representing a unique opportunity to restore processes of riverine wetlands that have been drastically reduced since the mid-19th century due to land reclamation, hydraulic regulation, and environmental degradation (Viaroli et al., 1996; Santoul et al., 2004; Lenda et al., 2012). In order to estimate the potential of PLs to play a key role in the riverscape rehabilitation and to provide ecosystem services, it is necessary to obtain information on the actual number of PLs, their distribution, their shape and similarity to natural W&Os, and the anthropogenic pressure acting on them. In this regard, limnological methodologies that rely on land-based measurements to study lentic water bodies on a basin-scale can be expensive and time-consuming. On the other hand, satellite remote sensing offers reliable tools for large-scale and repeated surveys of aquatic environments (Ghirardi et al., 2020, 2022; Free et al., 2021; Matta et al., 2022).

Given these premises, the present study aims to evaluate whether PLs can be exploited to replace the lost or degraded W&Os for restoring processes and functions in the Po River basin floodplain. To achieve this aim, a synergistic approach was used including remote sensing techniques, satellite archives, and regional databases.

2.2 Materials and methods

2.2.1 *Study area*

The Po River is one of the largest rivers in the Mediterranean Sea. Its watershed is ~74,000 km^2 wide, of which ~71,000 km^2 in Italy. The main hydrographic network is composed by the Po River (652 km length) and 141 major tributaries (6750 km total length). Countless natural and artificial channels also form a secondary hydrographic network with ~31,000 km total length (Montanari, 2012) (Fig. 1). The Po River basin represents a key resource for the Italian economy. Specifically, livestock breeding, agriculture and industry account for 55%, 35% and 37% of the sector GDP, respectively (Viaroli et al., 2010; Musolino et al., 2017). The Po River basin contributes for about 60% of national sand and gravel production. A remarkable change of this sector occurred from mid-1950's till 1980's, when the annual aggregate extraction from river beds in the Po River basin increased exponentially from 0.5x10^6 m^3 (1955) to 12.0x10^6 m^3 (1975) to support post-war reconstruction and the subsequent economic development (Lamberti, 1993). Due to the combined effects of aggregate removal, river damming and embankment, a riverbed incision of about 5 m was detected from 1960 to 2000 in the middle reach of the Po River, downstream of Cremona (Rinaldi et al., 2010). Once the dramatic alteration of the river morphology was perceived, sand and gravel withdrawal was moved to the floodplain. Indeed, aggregate quarries in the floodplain were also exploited in the 1960s in the Turin and Milan surroundings, where an unprecedent urban growth occurred, creating many shallow lakes (Di Natale and Durio, 1985).
In terms of water resources, water from the Po River is overexploited for irrigation, hydropower generation, and domestic purposes (Montanari, 2012; Coppola et al., 2014). Water availability, as well as water management and storage, show considerable differences among the northern and southern sides of the river. The northern basin area is characterised by a great water availability due to the presence of four deep subalpine lakes fed by Alpine glaciers (Lakes Maggiore, Como, Iseo, and Garda), and some hundreds of small natural and artificial lakes. By contrast, the southern part of the basin is affected by water scarcity with streams and rivers recently becoming intermittent, with frequent drought episodes especially in summer during the last two decades (Po River Watershed Authority, 2016).

Figure 1. True colour representation of the Po River basin divided into hydroecoregions (HER). The dark blue line represents the Italian borders of the basin, while the light blue line identifies the Po River. The yellow lines represent the hydroecoregions (HERs): Cod. 1 (Inner Alps), Cod. 2 (Inner Alps-E), Cod. 3 (Inner Alps-S), Cod. 11 (Calcareous southern Alps and Dolomites), Cod. 18 (Apennines-N), Cod. 21 (Ligurian Alps), Cod. 24 (Piemonte Apennines), Cod. 71 (Monferrato), Cod. 132 (Po Plain). The map on the upper right represents the location of the basin nationwide.

2.2.2 *Identification and classification of water bodies in the Po River basin*

A synergistic approach between remote sensing techniques and regional hydrographic databases was used to identify all lentic water bodies in the Po River basin. First, twelve Sentinel-2 (S2) Level 2 images (L2, atmospherically corrected) free of clouds covering the entire basin were selected and downloaded for the end of 2021. Specifically, the dates chosen were the 5th and 10th of September 2021, and 15th, 16th, 18th and 28th of October 2021 (Fig. 2). The S2 mission consists of twin polar, multispectral, high spatial resolution satellites (S2A and S2B) placed on the same orbit but offset 180° from each other. This feature allows a revisit time of about 5 days. Located aboard these satellites is the MultiSpectral Instrument (MSI) sensor characterised by 13 spectral bands (442-2202 nm) with three different spatial resolutions (10, 20 and 60 m). The S2 images were downloaded from the ONDA portal (https://catalogue.onda-dias.eu/catalogue/) and resampled to the same spatial resolution (10 m) using the "Resampled" tool of the SNAP (SeNtinel Application Platform) software. To identify all aquatic pixels in each image, two spectral indices were applied: WAVI (Water Adjusted Vegetation Index) (Eq. 1) and RWI (Relative Water Index) (Eq. 2). The WAVI was proposed by Villa et al., (2014a) and was specifically designed to capture aquatic vegetation features. This index was tested on different sensors (including the MSI of S2) and several study sites and showed high sensitivity to vegetation features and excellent performance in distinguishing aquatic from terrestrial vegetation (Villa et al., 2014a, 2014b, 2015, 2020). In our case study, this index is reliable because it can efficiently distinguish aquatic pixels (negative values) from everything else in the image. However, as the S2 images are time-staggered (September-October 2021), it was not advisable to use a constant threshold, so an adaptive threshold was used for each image based on the intensity of the spectral reflectances.

The RWI index was proposed by Wu et al. (2020) and its value ranges from -1 to +1 just like the Normalized Difference Water Index (NDWI), with positive values indicating aquatic pixels and values between 0 and -1 representing vegetation and bare soil. Although the NDWI index is widely used (e.g., Du et al., 2016; Kaplan and Avdan, 2017; Yang et al., 2017; Sekertekin et al., 2018) it has been proven that its threshold can vary greatly over time and be highly subjective (Ji et al., 2009); consequently, we selected the RWI index. Furthermore, in the work of Wang et al. (2021) it was observed that the RWI index can efficiently remove shadows projected by mountains, buildings, and clouds, as well as discard so-called mixed pixels, i.e., all those pixels located in the water-ground interface.

Eq. 1 $WAVI = (1 + L)\frac{B8 - B2}{B8 + B2 + L}$

Eq. 2 $RWI = \frac{(B3 + B5) - (B8 + B8A) + B12}{B3 + B5 + B8 + B8A + B12}$

Where B2 is the blue band (central wavelength = 490 nm), B3 is the green band (central wavelength = 560 nm), B5 is the vegetation red edge band (central wavelength = 705 nm), B8 is the NIR band (Near Infrared Reflectance, central wavelength = 842 nm), B8A is another vegetation red edge band (central wavelength = 865 nm), B12 is the SWIR band (Short Wave InfraRed, central wavelength = 2190 nm) and L is a correction factor that adjusts the influence of vegetation background (Huete, 1988).

Both indices were applied to the S2 images, and polygons representing all lentic water bodies were obtained from their combination. Then, to supplement the dataset, these polygons were compared and integrated with data downloaded from regional geoportals (digital archives containing geospatial data including water bodies). However, these databases are often incomplete and out of date, consequently, making the use of satellite imagery even more crucial.
Once the dataset including all water bodies was created, we removed all lakes of natural origin (e.g., glacial, barrage, or coastal lakes) except pristine riverine wetlands (e.g., oxbow lakes). All the remaining water bodies were then classified into nine macro categories based on origin, location and end use: agricultural ponds, aquaculture lakes, dam lakes, forest lakes (located within or near woodland areas), high-altitude lakes (above 1000 m), PLs, unclassifiable lakes (mainly because of their very small size), urban lakes (located within parks or close to urban and/or industrial areas) and W&Os. The exclusion of lakes of natural origin and the classification of water bodies was based on the expert judgement of the operator. Specifically, the identification of the active PLs, was based on their regular shape and the presence of sand/gravel piles and excavation machinery (e.g., dredgers) in the surrounding areas; while the identification of the ceased PLs was based on online research, detection of evident ground modifications in the surrounding area, as well as their regular shape. Furthermore, based on a Landsat image archive (https://global-surface-water.appspot.com/) that includes three million satellite images from 1984 to 2016 (spatial resolution of 30 m), the start and the presumed end of mining activity of many of these lakes was estimated. The Landsat constellation is a satellite system developed through a USGS/NASA partnership (United States Geological Survey/National Aeronautics and Space Administration). In contrast, the identification of W&Os was based on their location (almost always in the immediate proximity of a waterway) and their shape: irregular and sinuous for wetlands and with the typical "U" shape for riverine oxbows.
For all the identified PLs within the Po River basin, more detailed analyses were carried out involving morphological and morphometric characterization. Specifically, polygons were created

using QGIS and applied to S2 images to calculate perimeter, area and sinuosity index (D_L) of PLs. The latter parameter represents the articulation of the shoreline and is calculated by comparing the perimeter of a lake with the perimeter of a circle having the same area (Eq. 3): the closer the value of D_L is to 1, the less sinuous the lake is and the closer it is to a circular shape (Tonolli, 1964).

Eq. 3 $D_L = \frac{P}{2\sqrt{\pi A}}$

Where "P" represents the perimeter and "A" the area of the water bodies. Specifically, this morphometric analysis was conducted on PLs throughout the entire basin.

2.2.3 *Spatio-temporal analysis of PLs and W&Os*

More in-depth analyses were conducted in a subsample of eight geographic areas (Fig. 2): Turin (TO), Po and Orba River Park (OR), Milan (MI), Trezzo sull'Adda (TR), Brescia (BS), Mantua (MN), Modena (MO), and along the main Po River shaft (PO). These areas were selected because they are environmentally interesting and each is representative of a different land use. In each of these geographic areas, the PLs exhibit a high degree of heterogeneity and they well represent the entire basin. A numerical and morphological (perimeter, area, and sinuosity index) comparison between PLs and W&Os was performed for 2021. In addition, two temporal analyses were conducted: one concerning the evolution of the number and area of PLs, while the other focused on land use around PLs.

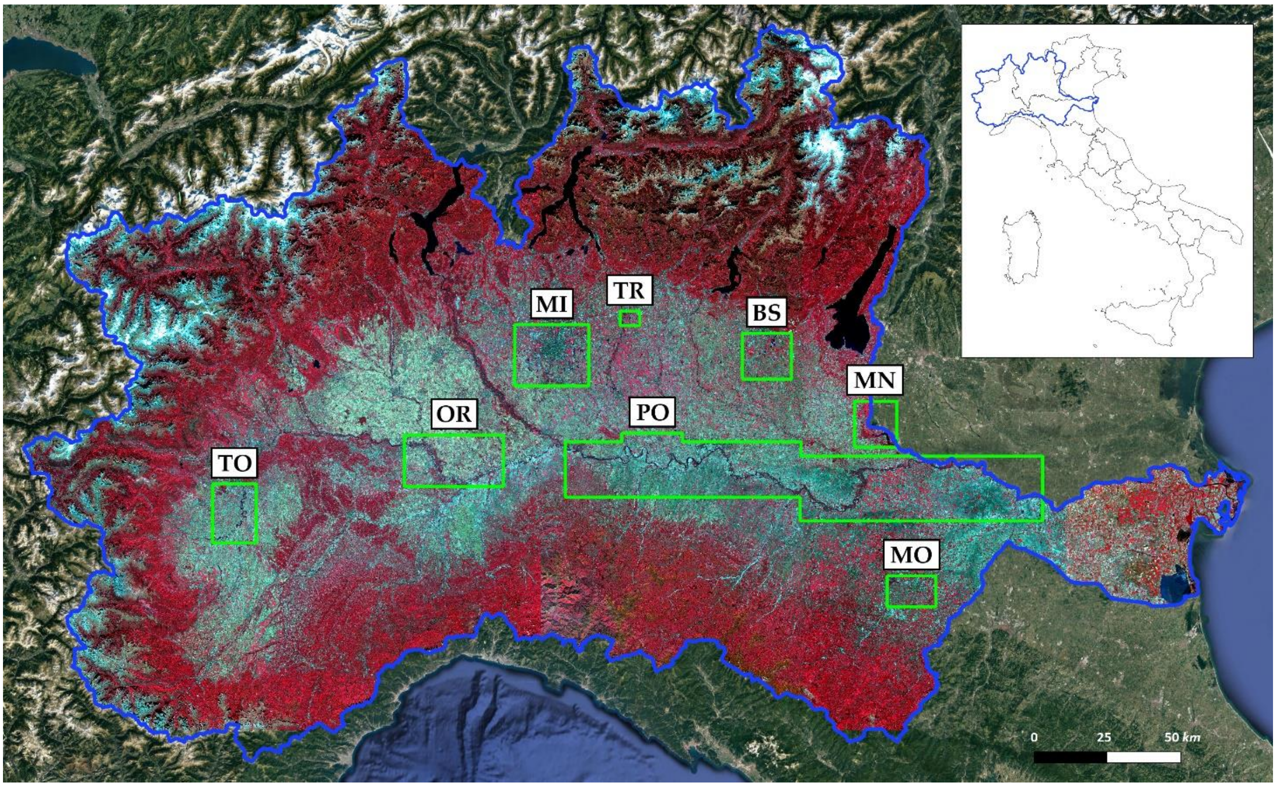

Figure 2. Mosaic of Sentinel-2 satellite images from late 2021 displayed in CIR colour (Color InfraRed) covering the entire Po River basin (05/09, 10/09, 15/10, 16/10, 18/10 and 28/10). Green polygons delineate the eight areas of the subsample: Turin (TO), Po and Orba River Park (OR), Milan (MI), Trezzo sull'Adda (TR), Brescia (BS), Mantua (MN), Modena (MO), Po River shaft (PO). The map on the upper right represents the location of the basin nationwide.

In order to assess the temporal evolution of PLs number and total area, autumn S2 images of 2021 were compared with Landsat images of the same period from previous years: 14th September and

23rd October 2014 (Landsat-8, L8); 8th, 15th and 22nd September 2000 (Landsat-7, L7); 10th and 12th of September 1990 (Landsat-5, L5). These three satellites are characterised by a spatial resolution of 30 m and a revisit time of 16 days. The L5 and L7 carry on-board Thematic Mapper (TM) and Enhanced Thematic Mapper (ETM+) sensors characterised by 4 bands in the Visible-Near Infrared (VIS-NIR), while the Operational Land Imager mounted on the L8 is equipped 5 bands in the same spectral domain.
In order to assess the temporal evolution of land use around PLs, two types of buffer zones were used depending on the location of the water body and the pressures it is subjected to (Rossi et al., 2007): 500 m within the Po floodplain area and 250 m outside it. Land-use data were downloaded from the geoportals of the three main regions of the Po River basin: Piedmont (available years: 1990, 2000, 2006, 2012, 2018), Lombardy (available years: 1980, 1999, 2007, 2012, 2015, 2018) and Emilia-Romagna (available years: 1976, 1994, 2003, 2008, 2014, 2017) and were categorized following the first-level criteria of the Corine Land Cover (CLC) (CORINE Land Cover - Copernicus Land Monitoring Service): artificial surfaces (class 1), agricultural areas (class 2), forest and seminatural areas (class 3), wetlands (class 4) and water bodies (class 5).
Finally, within the same buffer zones, riparian vegetation around PLs was identified, differentiating wooded/shrubs areas (which could act as buffer zones for surface nutrient runoff) from cultivated farmland (sources of nutrients) using the Soil Adjusted Vegetation Index (SAVI), applied to the twelve S2 images of 2021. This index is similar to the NDVI index but applies a correction value to adjust the influence of vegetation background (Huete, 1988); as a result, the SAVI index works well even with low vegetation cover and allows the identification of fields with very different soils (Almutairi et al., 2013; Piragnolo et al., 2018; Qin et al., 2021) (Eq. 4).

Eq. 4 $SAVI = \frac{B8 - B4}{B8 + B4 + L} (1 + L)$

Where B4 is the red band (central wavelength = 665 nm) B8 is the NIR band (central wavelength = 842 nm) and L is the correction factor, which varies from 0 to 1 and depends on the intensity of green vegetation cover (L=1, areas with no green vegetation cover; L=0, areas with high green vegetation cover, i.e., equivalent to the NDVI method). In our case, the value of L is not constant, but was recalibrated according to the characteristics of each image, i.e., from the amount of green vegetation areas estimated by visual interpretation (L=0.8 for areas with low green vegetation cover, L=0.5 for areas with medium green vegetation cover, and L=0.2 for areas with high green vegetation cover).

In addition to the SAVI index, within the same buffer zones, a supervised classification was performed using the “maximum likelihood” tool, one of the most popular classification methods in remote sensing (Pham et al., 2019; Ha et al., 2020). This classification is based on the spectral signature of different surface types (Fig. 3). In detail, Regions of Interest (ROIs) were created including pixels of each surface: water bodies, sand, cultivated farmland, bare soils, urban structures (based on *in situ* observations conducted at ground level between September and October 2021) and riparian vegetation (identified from *in situ* and SAVI index products, useful to increase the spatial ability to detect this surface type). Once the software is “educated” through these ROIs, it can automatically recognize the different types of surfaces in all buffer zones. Figure 3 shows some examples of spectral signatures (reflectances) obtained from S2 images and documents how the spectral indices are able to distinguish different surfaces. The WAVI index used the B2 and B8 bands, and the only surface that has B2>B8 is water, so the water pixels will be the only negative ones. Similarly, the SAVI index uses the B4 and B8 bands and due to the difference in the reflectance value of these two bands is able to distinguish the various surfaces.

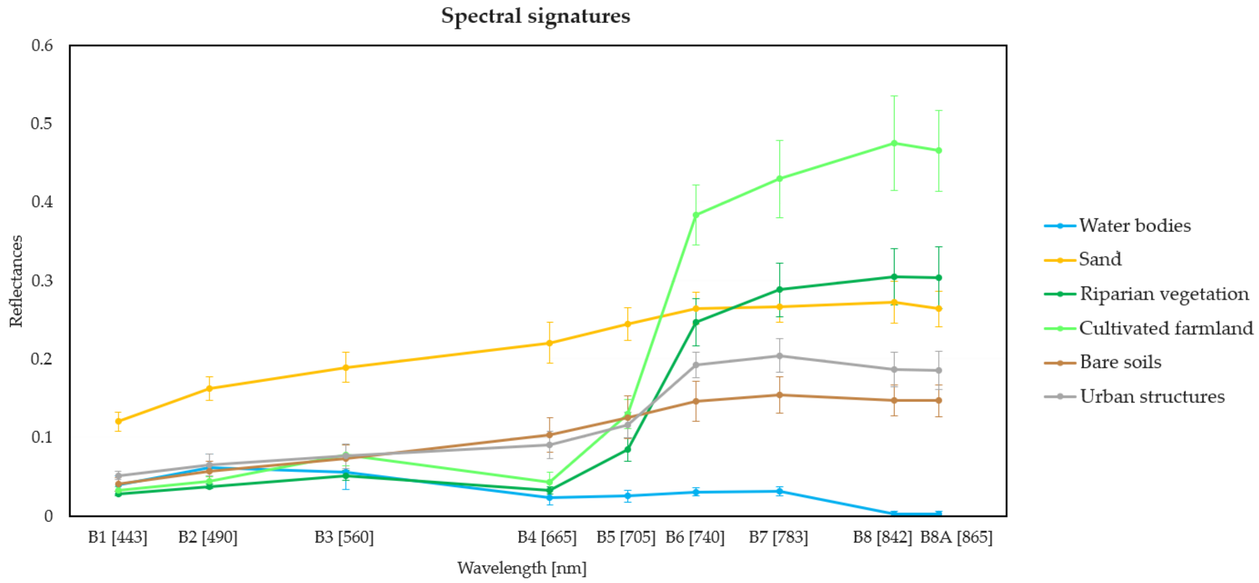

Figure 3. Example of spectral signatures (reflectances) obtained from Sentinel-2 images for different types of surfaces used for the "maximum likelihood" classification.

2.3 Results

2.3.1 *Identification of water bodies in Po River basin*

The comparison between data available in the regional geoportals and the polygons obtained by applying WAVI and RWI indices to S2 images, allowed us to identify 12,685 small lentic water bodies in the entire Po River basin. The combined approach of the two spectral indices was proven useful and reliable for the identification of all water bodies. WAVI index was able to accurately identify all water pixels, while RWI index allowed removal of all mixed pixels at the land-water interface. The water bodies identified in the Po River basin were then classified into nine macro-categories according to origin, location, and destination use (Fig. 4).

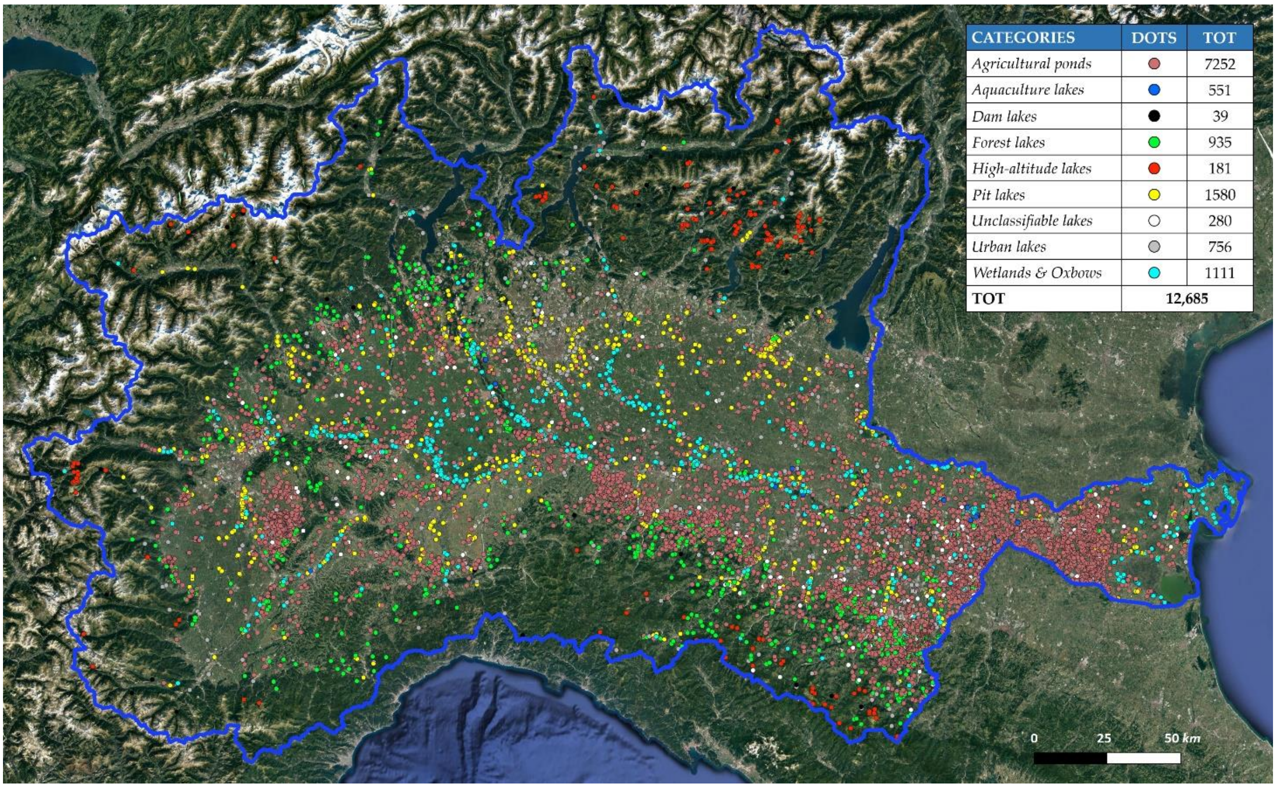

Figure 4. Water bodies of the Po River basin divided into nine macro categories: agricultural ponds (brown), aquaculture lakes (blue), dam lakes (black), forest lakes (green), high-altitude lakes (red), pit lakes (yellow), unclassifiable lakes (white), urban lakes (gray) and wetlands & oxbows (light blue).

Unclassifiable lakes represent all those water bodies that are difficult to classify within the macro-categories due to their small size and/or inability to classify their end use. Overall, the most represented category is “agricultural ponds”, followed respectively by “pit lakes” (PLs) and “wetlands & oxbows” (W&Os). The “agricultural ponds” category consists of lakes located near croplands and used primarily for irrigation (6654) and water bodies inside farms that are used for wastewater storage (598). The “urban lakes” category consists of all water bodies situated inside or in proximity of urban agglomerations (367), in urban parks (57), inside golf courses (188) or near industrial areas (144). Finally, the “pit lakes” category was divided into three subcategories: active PLs (338), ceased PLs (1188), and doubtful PLs (54, i.e., all those lakes that have the typical characteristics of pit lakes but whose origin or end of mining is uncertain).

We also evaluated the distribution of small lentic water bodies in the hydro-ecoregions (HERs) of the watershed (Tab. S1). The HER with the largest number of small lentic water bodies is the “Po plain” (Cod. 132), with a density of 0.34 water bodies per km^2. More generally, HERs that are more extensive and with a lower average elevation are characterised by a higher density of water bodies. Focusing exclusively on PLs, the number of active, ceased and doubtful PLs in the Po River basin are reported in Fig. 5. Details on the distribution of PLs within HERs are also reported in Table S2.

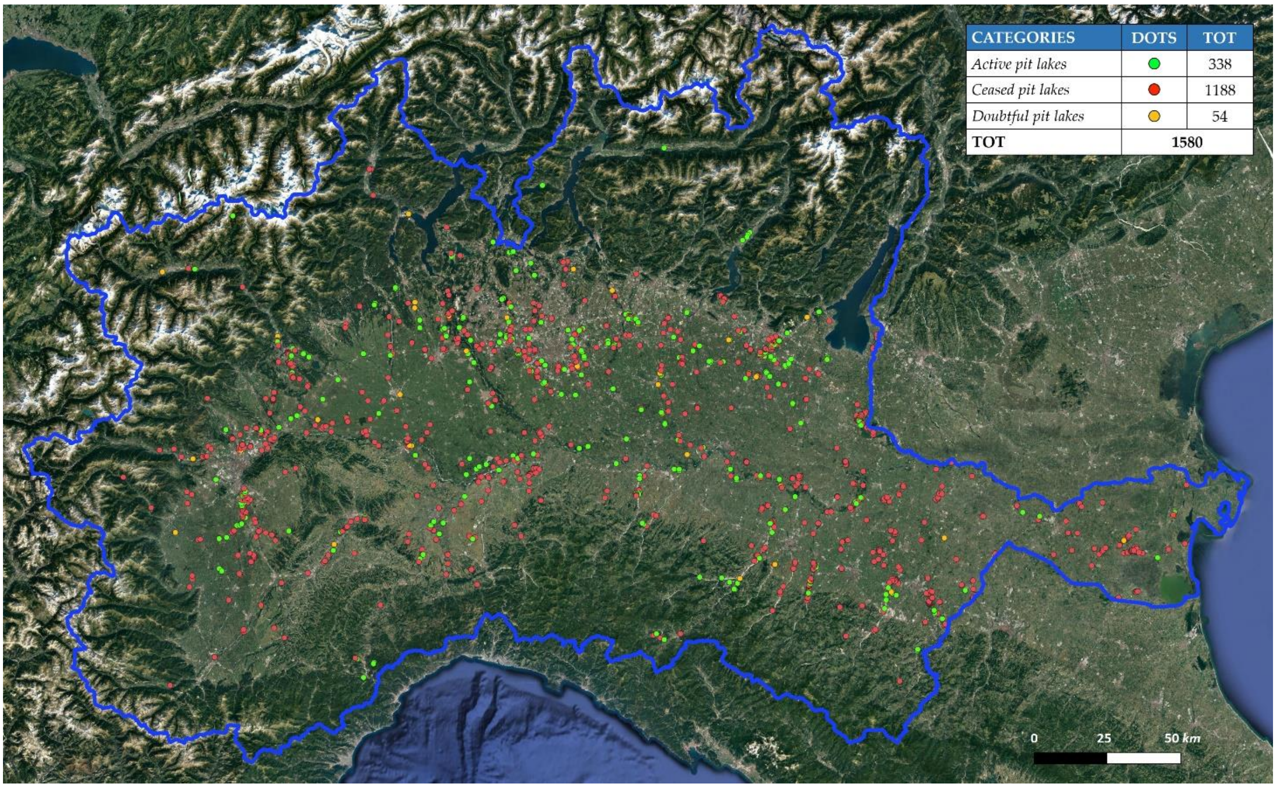

Figure 5. Pit lakes of the Po River basin divided into three subcategories based on mining activity: active (green), ceased (red) and doubtful (yellow).

2.3.2 *Morphometric analysis*

The total area of all PLs in the Po river basin (63.5 km^2) is almost equivalent to the area of Lake Iseo (65.3 km^2), one of the large perialpine lakes, while the total perimeter (1319 km) is about twice the length of the Po River. The main morphometric features, i.e., perimeter, area and sinuosity index are reported in Figure 6. Nearly 92% of the PLs are characterised by a perimeter <2000 m. More than one-third of PLs are small with area <1 ha, due to the quarry management during the early phase of the quarry exploitation. We also identified a high number of PLs (308) with area 5-25 ha and 41 PLs with area >25 ha. Most of the PLs exhibit a rather regular shape, as 91% of them are characterised by a low sinuosity, with $1<D_L<1.8$ and mean $D_L=1.39$ (± 0.35).

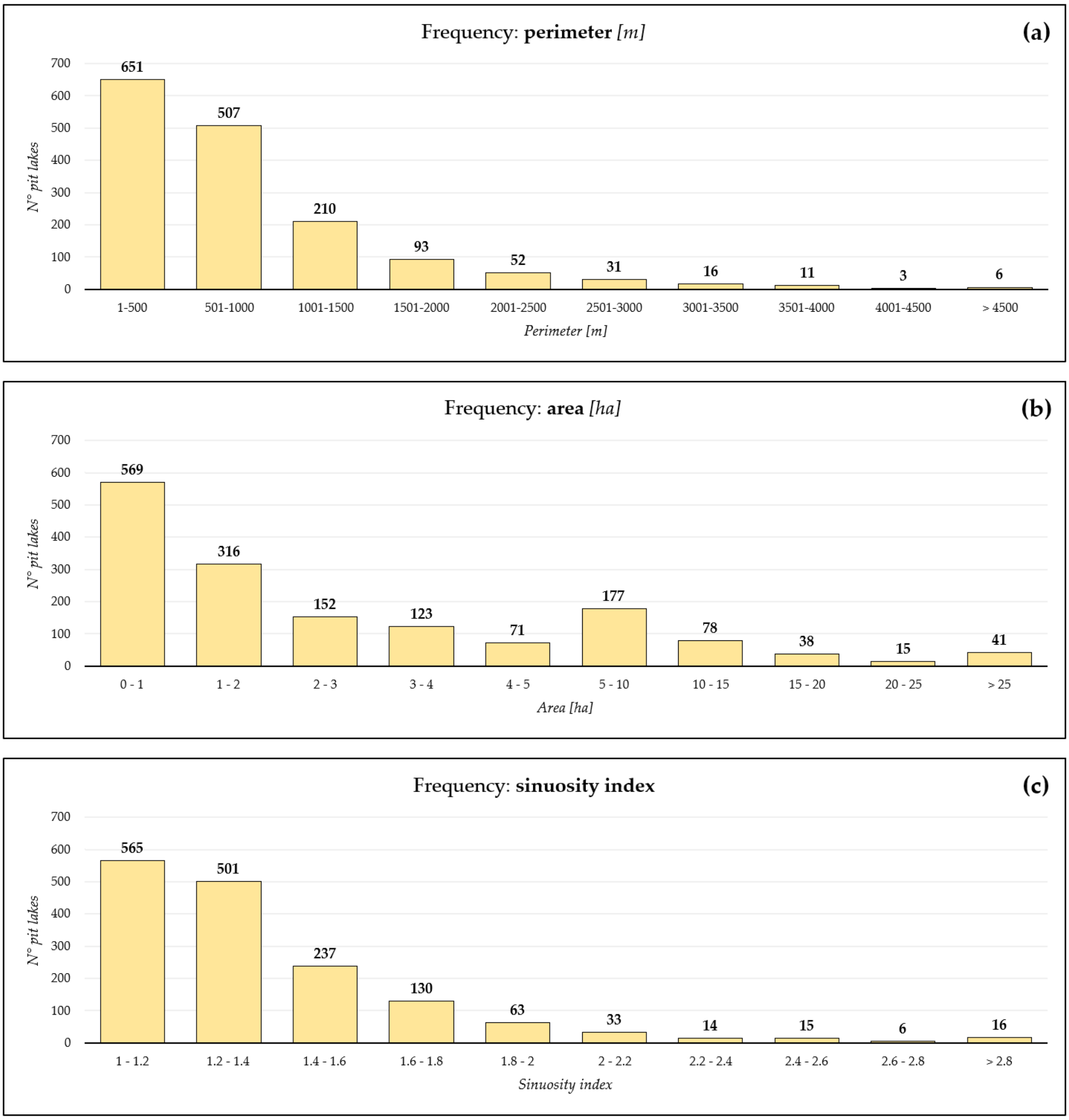

Figure 6. Distribution of the three morphometric indices of pit lakes in the Po River basin in 2021. Perimeter (a), area (b) and sinuosity index (c).

The same morphometric parameters (perimeter, area and sinuosity index) were also calculated within the eight geographic subsample areas, with a focus on the comparison between PLs and W&Os. PLs (320) are more numerous than W&Os (238) (Fig. 7). PLs are evenly distributed in all subsample areas, while 79% of W&Os are concentrated in only two areas (OR and PO; 0.07 W&Os km^{-2} compared to 0.02 W&Os km^{-2} of the others subsample areas). The total area (2980 ha) and total perimeter (461 km) of PLs are much greater than those of W&Os (453 ha and 248 km, respectively) (Tab. 1). However, due to their narrow and elongated shape, the sinuosity index of these natural water bodies (2.2 ± 1.1) is greater than that of PLs (1.4 ± 0.3). Nonetheless, PLs can provide a land-water interface which is twice that of W&Os, i.e., a high biogeochemically reactive zone.

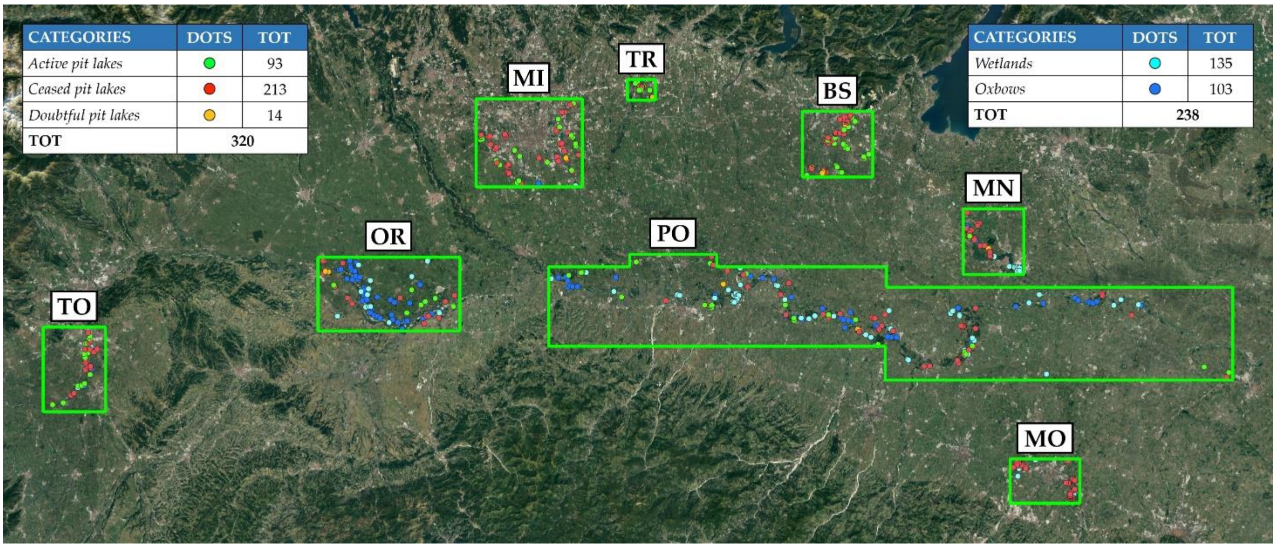

Figure 7. Pit lakes (PLs) and wetlands & oxbows (W&Os) located within the subsample areas. Active PLs (green dots), ceased PLs (red dots), doubtful PLs (yellow dots), wetlands (light blue dots) and oxbows (dark blue dots). Turin (TO), Po and Orba River Park (OR), Milan (MI), Trezzo sull'Adda (TR), Brescia (BS), Mantua (MN), Modena (MO), Po River shaft (PO).

Table 1. Numerical and morphometric comparison between pit lakes (PLs) and wetlands & oxbows (W&Os) within the subsample areas in 2021. TO (Turin), OR (Po and Orba River Park), MI (Milan), TR (Trezzo sull'Adda), BS (Brescia), MN (Mantua), MO (Modena) and PO (Po River shaft).

	Count *[n]*		**Area Tot.** *[ha]*		**Perimeter Tot.** *[km]*		D_L *[mean ± st.dev.]*	
	PLs	W&Os	PLs	W&Os	PLs	W&Os	PLs	W&Os
TO	33	5	441	12	59	6	1.4 ± 0.3	2.2 ± 1.1
OR	25	64	215	100	40	72		
MI	56	4	633	1	88	1		
TR	13	-	91	-	15	-		
BS	58	-	463	-	71	-		
MN	24	27	123	54	25	24		
MO	41	13	281	18	46	7		
PO	70	125	735	267	117	137		
Tot.	**320**	**238**	**2980**	**453**	**461**	**248**		

2.3.3 *Temporal evolution of pit lakes*

The temporal evolution analysis performed on the PLs subsample showed that the number and the area of lakes increased between 1990 and 2021, but with different trends among the subsample areas (Fig. 8; Tab. S3). An example of temporal evolution concerning some PLs located in the PO area is shown in Figure S2.

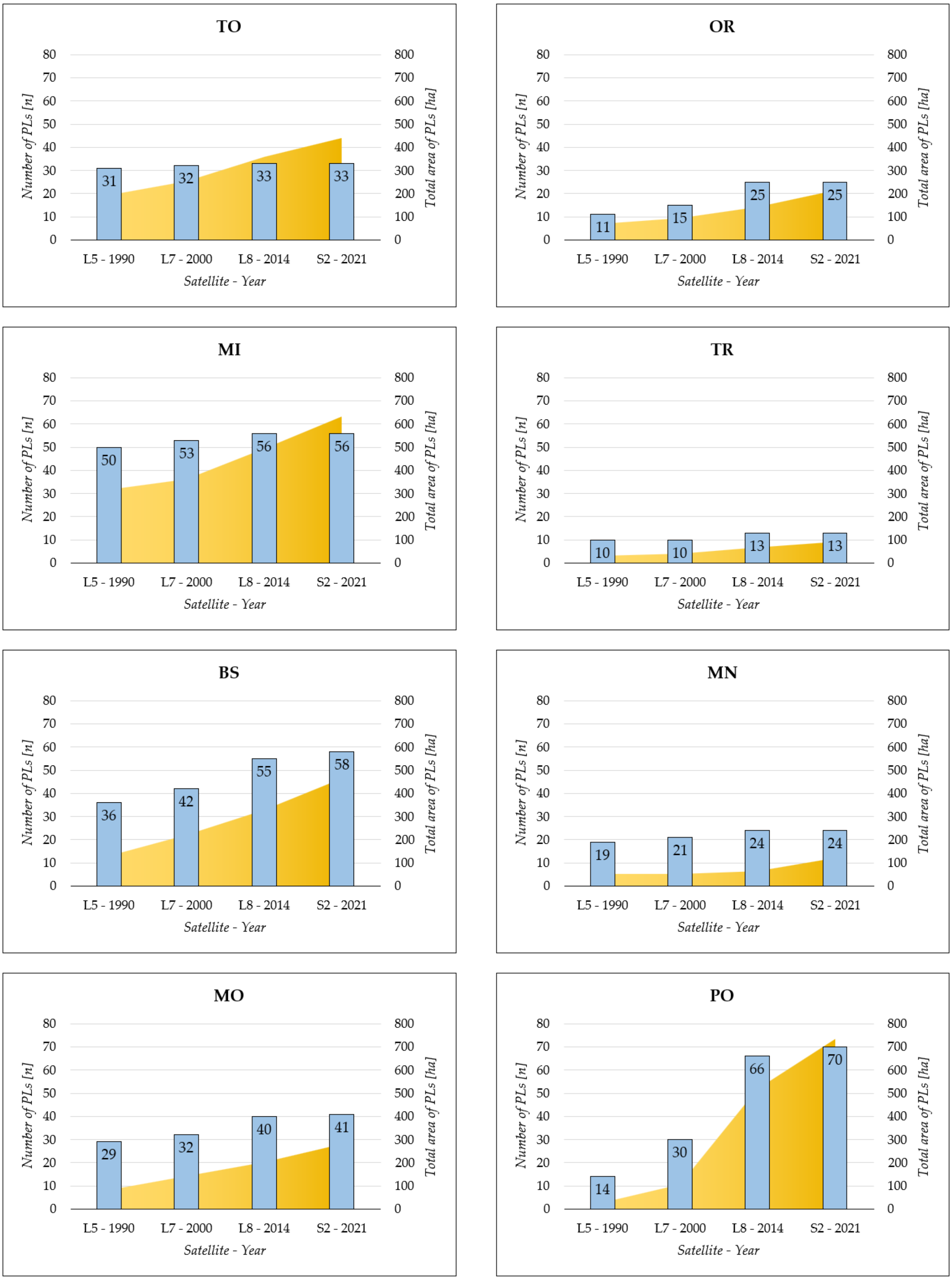

Figure 8. Temporal evolution of number and area of pit lakes (PLs) in the subsample areas. The blue columns represent the number of PLs while the orange sections represent the total area. TO (Turin), OR (Po and Orba River Park), MI (Milan), TR (Trezzo sull'Adda), BS (Brescia), MN (Mantua), MO (Modena) and PO (Po River shaft). Landsat-5 (L5), Landsat-7 (L7), Landsat-8 (L8) and Sentinel-2 (S2).

In the areas close to urban zones (TO and MI, and to a lesser extent TR, MO, BS and MN), the PLs number was very high before 1990 and did not grow further in the next decades. By contrast, in the other areas there was a progressive growth of both PLs number and size. This pattern is clearly evident for the two main riverine areas (OR and PO), where PLs appeared after 1990, when aggregate withdrawal was moved from the riverbeds to the floodplain and beyond. In the PO area, also the areal extent underwent an exponential growth, from 28 ha in 1990 to 735 ha in 2021. A slight decrease of quarry exploitation was detected after 2014, following the world-wide economic crisis.

2.3.4 *Land use analysis around pit lakes*

From 1990 to date, the land use in the buffer zones around the PLs has seen a decrease in agricultural areas (CLC class 2) in favour of urban agglomerations (CLC class 1) and wooded and semi-natural areas (CLC class 3) in buffer zones around PLs. This trend is observed in all subsample areas and reflects the tendency visible for the whole Po River basin. The decrease in class 2 is well marked in all subsample areas, from a minimum of 11% (OR) to a maximum of 48% (TR). The largest increase in class 1 was found in the areas of MI (18%), TR (35%) and BS (31%), while it was more limited in the two areas close to the Po River: OR (5%) and PO (6%). On the other hand, the increase in class 3 was rather uniform across all subsample areas (10-12%). Despite the observed trend, class 2 still remains the dominant class, followed by class 1 and class 3 (except for TR where class 1 is the most represented).
The results of the classification of different surface types within lake buffer zones, from S2 images of 2021, are reported in Figure 9, for all the subsample areas. Excluding water pixels (PLs and rivers), the following surfaces were identified: quarry areas (sand), riparian vegetation, cultivated farmland, bare soils and urban structures. This analysis improves the results obtained from regional land-use data, partially confirming the results already obtained (e.g., the dominance of agricultural areas and the high presence of urban agglomerations in MI, TR, and BS areas). The extent of riparian vegetation is not homogeneous within the subsample areas and often depends on quarrying activities. Around active PLs the soil is either unvegetated or with ruderal and sparse vegetation along the littoral zone (median: 23%), while around ceased PLs the littoral vegetation is denser and more continuous, often attaining homogeneous coverages along the entire perimeter of the lakes (median: 34%). As a result, in areas with many active PLs such as TO and BS, the percentage of riparian vegetation in buffer zones is low (22% and 23%, respectively), while in areas where all mining has ceased such as MO, the percentage is significantly higher (44%). The small percentage found in the MN area, despite low mining activity, is instead due to the agriculture spreading over 71% of the buffer zone (Tab. S4).

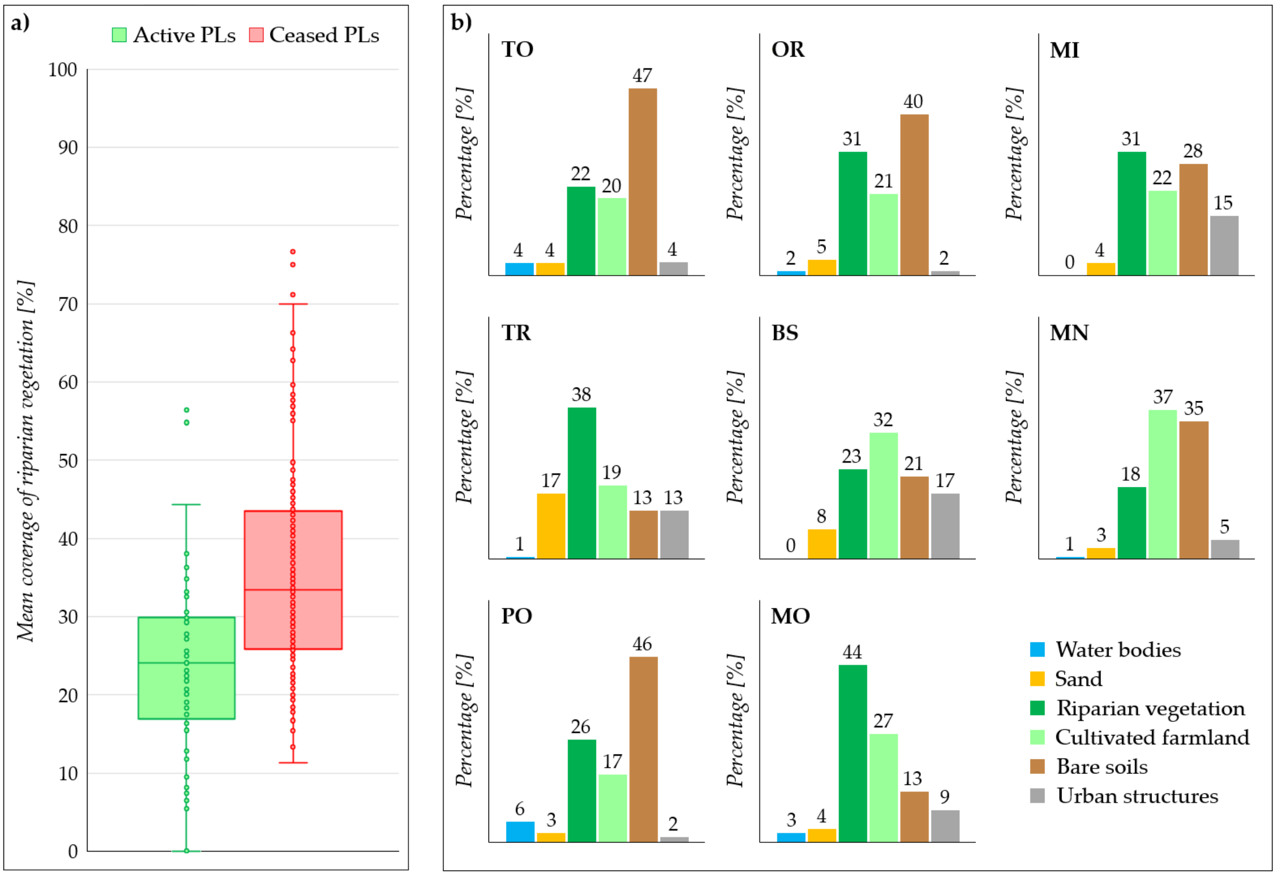

Figure 9. a) comparison of riparian vegetation cover in the buffer zone around active (green) and ceased (red) pit lakes. B) maximum likelihood classification in pit lakes buffer zones into the subsample areas. TO (Turin), OR (Po and Orba River Park), MI (Milan), TR (Trezzo sull'Adda), BS (Brescia), MN (Mantua), MO (Modena) and PO (Po River shaft).

2.4 Discussion

2.4.1 *Reliability of the remote sensing to identify small lentic water bodies*

In this study we assessed the extent of small lentic water bodies in the Po River watershed (Northern Italy) with remote sensing. Specifically, we focussed on PLs by detecting their number, distribution and main morphometric features along with their temporal changes. The land use in the surroundings of the lakes was also evaluated. The temporal evolution of land use revealed a general decrease in agricultural areas in the Po River basin, in favour of urban structures and wooded and semi-natural areas. This trend can be observed both within the subsample areas and in the buffer zones around the PLs. Nevertheless, agricultural areas remain the dominant land use class in this basin.

This study appears to be a novel in the context of spatio-temporal analysis of PLs at basin-scale, while there are several works concerning the identification of small water bodies in different parts of the world (Olmanson et al., 2008; Khandelwal et al., 2022; Pi et al., 2022). The Sentinel-2 images, with a spatial resolution of 10 m and a revisit time of about 5 days, are recognized as appropriate for the census and the classification of relevant water bodies in the Po River basin. In detail, the combined use of the two spectral indices WAVI and RWI allowed the accurate detection of all water pixels, avoiding mixed pixels located at the land-water interface. Likely, this is an innovation of this study since we do not find such an application in the updated literature on this topic. To date, the unsupervised classification (Olmanson et al., 2008) or the Landsat dataset "Global Surface Water Occurrence" (Khandelwal et al., 2022; Pi et al., 2022) were used instead, which are less accurate and reliable, often resulting in a misclassification. The application of the Landsat dataset is also biased by the coarse spatial resolution of the constellation sensors (30 m), while the 10-m spatial resolution of Sentinel-2 allows a much better identification and classification of water bodies >100 m^2. Integration of satellite imagery from the Landsat constellation to complement Sentinel-2 images from 1990 to 2021 allowed us to detect an increase in both number and areas of PLs especially in subsample areas OR and PO. In fact, there are consistent sand deposits in these areas that have been exploited since late 1980s, when the quarrying activity was moved from riverbeds to floodplains. This trend supports speculation that if sand quarrying will be managed also with ecological criteria it will provide an unprecedented opportunity to rehabilitate processes and functions of the pristine riverscape with PLs taking the place of W&Os.

2.4.2 *Gravel and sand pit lakes can be exploited for riverscape rehabilitation*

The classification of all the small lentic waters typologies in the Po River basin allowed the comparison between the extent of PLs and W&Os. PLs were more numerous and more widely distributed than W&Os. A high density of PLs and W&Os was detected in the floodplain of the lowland reach of the Po River (subsample area PO), which is heavily exploited for sand provisioning. By contrast, in the most urbanized zones (e.g., TO, MI, TR and BS), PLs are almost the only lentic water bodies. For these reasons, we hypothesized PLs can be exploited and managed as providers of ecosystem services previously provided by W&Os in the riverscape. The PLs and W&Os have similar morphometric characteristics, although PLs are generally wider than W&Os (PLs with area >5 ha are 349, while W&Os with area >5 ha are only 39). By contrast, PLs have a lower sinuosity than W&Os, i.e., the shallow littoral zone is less developed in PLs. This littoral area of the lakes can be considered as valuable habitats for biodiversity conservation as well as a "hot spot" of benthic processes contributing to water oxygenation, and nitrogen and phosphorus removal from water (Bruesewitz et al., 2012; Cross et al., 2014; Rodrigo et al., 2015; Nizzoli et al., 2018, 2020). Between 30 and 95% of total nitrogen removal may be due to the littoral benthic zone, however PLs are often designed to maximize cost-effectiveness, resulting in steep slopes and a high depth-to-surface area ratio (Kattner et al., 2000; McCullough and Lund, 2006; Blanchette and Lund, 2016; Søndergaard et al., 2018). Consequently, shallow waters are usually limited to a narrow strip along the land-water edges, representing only a small fraction of the total area, e.g., <20% (Nizzoli et al., 2020).

2.4.3 *Gravel and sand pit lakes can provide ecosystem services*

We hypothesized the PLs can be exploited for the restoration of some biogeochemical processes, assuming they can provide ecosystem services. This goal should be included in the quarry design in order to empower the most critical ecosystem components, e.g., shallow water habitats (Blanchette and Lund, 2016). However, after the cessation of the quarrying activities, PLs can spontaneously develop towards healthy aquatic systems that can be managed for rehabilitating aquatic habitats and aquatic ecological networks (Rossi et al., 2007). For example, the shallow littoral zone can be progressively colonized by a structured community of aquatic macrophytes which provides substrate, food and shelter for aquatic animals, and affects water and sediment chemistry, primary productivity and microbial transformations (Nizzoli et al., 2014; Søndergaard et al., 2018; Seelen et al., 2021). Therefore, mature PLs can achieve structure and functioning similar to those of natural water bodies such as oxbow and riverine lakes, thus providing the same key processes and ecosystem services (King et al., 1974; Tavernini et al, 2009a,b; Sienkiewicz and Gąsiorowski, 2016; Søndergaard et al., 2018). Among others, we consider here two typologies of

ecosystem service which address serious environmental issues, namely water provisioning in a drought scenario and nitrogen removal from N-contaminated waters.
As highlighted by Baronetti et al., (2020), the Po River basin has experienced several drought events since 1980s, along with an increase of ~2 °C of the mean annual temperature (Musolino et al., 2017). The most impactful events occurred in 2003-2007, 2012, 2017 and the last but not least is still persisting since early 2022 through at least 2023. Moreover, the frequency and intensity of drought events are expected to intensify in the next decades (Palatella et al., 2010), due to decreased precipitation during critical periods (spring-summer) along with an intensification of storms with short durations and high intensities in late spring and in summer (Diodato et al., 2020). In this context, the water storage in small lentic water bodies might provide at least a transient buffer against drought, whose extent will depend upon the stored volume. We tentatively estimated the water volume of all PLs in the watershed with a power regression model: $V=\alpha A^{\beta}$ (for details see Tab. S5 and Fig. S1). We used the coefficients α and β for small lakes in a riverine network (Izmailova and Koornenkova, 2022), lakes with $A<1.25$ km^2 (Delaney et al., 2022) and lakes with $A<0.30$ km^2 (this study, Nizzoli et al., 2020) obtaining 341×10^6 m^3, 359×10^6 m^3 and 458×10^6 m^3 respectively, with an average of 378×10^6 m^3. This water volume is nearly comparable with the total exploitable water volume (~1.0×10^9 m^3) in the main south-alpine lakes, i.e., L. Maggiore, L. Como, L. Iseo and L. Garda (www.laghi.net). We also focused on the subsample area PO, where there is a high number of both PLs (70) and W&Os (125), accounting for 7.35 km^2 and 2.67 km^2 surface area, respectively. This preliminary calculation does not consider that the water in the PLs would be a resource already present in the floodplain aquifers even if the pits were not excavated, but it does highlight that these environments could locally regulate the hydrological cycle and provide a readily accessible water storage. PLs in the floodplain are subject to the flood pulses (Junk et al., 1989), receiving and accumulating water during floods and releasing it gradually afterwards. This way, PLs can function as a retention system making the water available for days after the flood ceased. In the PO area, the water volume of PLs calculated as for the whole Po River basin ranges between 12.5 and 15.7×10^6 m^3 (mean = 13.7×10^6 m^3) which is relevant for the local hydrological cycle. Moreover, the water level pulses affect oxygen availability in both water column and top sediments, thus regulating the redox conditions and the main biogeochemical processes, e.g., those controlling the nitrogen pathways and fate (Racchetti et al., 2011; Bartoli et al., 2012, Nizzoli et al. 2020). Therefore, PLs account not only for water storage, but they behave also as metabolic reactors which could regulate biogeochemical processes in the floodplain (Wetzel, 1990).
We compared the performance of PLs and W&Os in terms of nitrogen removal based on the outcomes of previous studies in this area (Tab. S6). The total denitrification provided per unit area by the littoral zone of PLs (mean = 61 mg N m^{-2} d^{-1}, Nizzoli et al., 2020) is nearly comparable with the nitrogen removal from healthy riverine wetlands (mean = 114 mg N m^{-2} d^{-1}, Racchetti et al., 2011). These denitrification rates are among the highest in the literature (see Blevins, 2004; Racchetti et al., 2011; Bartoli et al., 2012; Nizzoli et al., 2014; 2020, and references therein) evidencing that shallow zones of PLs can perform as healthy W&Os in nitrogen removal. By contrast, the mean denitrification rates of both hypolimnetic PLs sediments (mean = 18 mg N m^{-2} d^{-1}) and degraded wetlands (mean = 16 mg N m^{-2} d^{-1}) were much less effective. From the data in Table S6, we tentatively estimated the nitrogen removal provided by the whole PLs assuming the littoral zone is 20% of the total lake surface area (Nizzoli et al., 2020). Namely, the littoral zone accounts for 12.7 km^2 area (LA) and 61 mg N m^{-2} d^{-1} denitrification rate (LD), and the hypolimnetic sediments contribute for 50.8 km^2 area (HA) and 18 mg N m^{-2} d^{-1} denitrification rate (HD). Thus, the mean nitrogen removal by whole lakes (MNR) is:

Eq. 5 $MNR = \frac{(LA * LD) + (HA * HD)}{(LA + HA)}$

Under these conditions, on average PLs can remove 26.6 mg N m^{-2} d^{-1} which is comparable to the N-surplus of 16.5 mg N m^{-2} d^{-1} estimated for the cropland in 2010 (Viaroli et al., 2018; Tab. 6S). These estimates highlight the potential of PLs to control excess N loads from agriculture and to act as nutrient filters. However, the nitrogen removal by denitrification depends on many factors, e.g., lake morphology, especially the extent of shallow waters versus the hypolimnetic, often anoxic, sediments, thermal regime, and trophic status (Nizzoli et al., 2010; 2014; 2020). These constraints can be addressed with a preventive inclusion of ecological criteria in the quarry design, e.g., wider littoral/shallow zones and marshes (see Rossi et al., 2007). Furthermore, to date the PLs surface area (63.5 km^2) is only 0.2% of cropland (30,600 km^2), namely it is almost negligible at the watershed scale, while it can be more effective locally, e.g., in the PO area due to the high density of PLs.

2.4.4 *Risk and threats*

Gravel and sand quarrying is far from problem-free; it has several negative effects differing in importance relative to the area and the climatic zone (Søndergaard et al., 2018). One of the major issues related to PLs is eutrophication, mainly due to surface runoff. It has been demonstrated how, in highly populated contexts with a high density of agricultural and urban areas, surface runoff and subsurface drainage can lead to rapid degradation of the trophic status of PLs (Sutton et al., 2011; Billen et al., 2013; Søndergaard et al., 2018), which can be associated with intense algal bloom (Cobelas et al., 1992; Codd, 2000; Tavernini et al., 2009a; Cross et al., 2014). In addition, PLs fed by rivers can become eutrophic or hypertrophic due to the high nutrient loads they transport (Kattner et al., 2000). Moreover, land-surface changes due to quarrying can affect the quantity and quality of groundwater, leading, for example, to contamination by chloride and nitrate (Welhan, 2001; Peckenham et al., 2009). Other effects of mining are the loss of soil protection (Rutherford et al. 1992; Kalbitz et al. 2000; Weilhartner et al., 2012) and the loss of freshwater due to evaporation (Mollema and Antonellini, 2016). Although these artificial aquatic systems represent a potential risk to the water table, Muellegger et al. (2013) demonstrated that PLs can improve groundwater quality. Specifically, the physical-chemical and biological processes within the lake and especially the filtration operated by the banks play a crucial role (Kedziorek et al., 2008; Massmann et al., 2008; Wiese et al., 2011). Surface runoff and the resulting discharge of nutrients and pollutants into the lake water could be substantially reduced in the presence of riparian vegetation around the perimeter of the lake, acting as a buffer zone. In this study, the SAVI index was used to semi-automatically detect all riparian vegetation areas around the PLs. Moreover, one of the most common supervised classification tools (maximum likelihood) was used to classify land use in the surroundings of PLs. This dual analysis revealed that the amount of riparian vegetation is strictly related to the quarrying activity, because once the quarry exploitation is concluded, the natural and gradual reforestation of the surrounding areas begins. The maximum likelihood classification tool is used frequently in the literature and it is suitable for multispectral satellite images (Ahmad and Quegan, 2012; Sisodia et al., 2014; Hossain et al., 2015; Pham et al., 2019; Karimzadeh et al., 2022). In relation to the spectral resolution, the future use of hyperspectral sensors could ensure a more accurate identification of riparian vegetation and a better land-use classification.

2.5 Supplementary materials

Table S1. Distribution of water bodies (WBs) into the hydroecoregions (HERs) that compose the Po River basin and divided into nine macro categories. Cod. 1 (Inner Alps), Cod. 2 (Inner Alps-E), Cod. 3 (Inner Alps-S), Cod. 11 (Calcareous southern Alps and Dolomites), Cod. 18 (Apennines-N), Cod. 21 (Ligurian Alps), Cod. 24 (Piemonte Apennines), Cod. 71 (Monferrato), Cod. 132 (Po Plain).

	Cod.1	**Cod.2**	**Cod.3**	**Cod.11**	**Cod.18**	**Cod.21**	**Cod.24**	**Cod.71**	**Cod.132**	**Tot.**
	8900 km^2	4763 km^2	4975 km^2	6663 km^2	8719 km^2	1225 km^2	1604 km^2	3033 km^2	29,906 km^2	69,788 km^2
Agricultural ponds	8	1	79	19	953	-	21	277	5894	**7252**
Aquaculture lakes	-	-	-	22	-	-	-	1	528	**551**
Dam lakes	13	7	6	5	6	-	-	1	1	**39**
Forest lakes	30	1	35	10	369	4	21	59	406	**935**
High-altitude lakes	10	12	28	91	38	2	-	-	-	**181**
Pit lakes	14	2	10	47	52	-	5	36	1414	**1580**
Unclassifiable lakes	-	-	-	-	45	-	-	26	209	**280**
Urban lakes	18	20	16	20	57	1	-	25	599	**756**
Wetlands & Oxbows	5	4	5	8	14	-	2	22	1051	**1111**
Tot.	**98**	**47**	**179**	**222**	**1534**	**7**	**49**	**447**	**10,102**	**12,685**
Density (WBs/km^2)	0.01	0.01	0.04	0.03	0.18	0.01	0.03	0.15	0.34	**0.18**

Table S2. Distribution of the three subcategories of pit lakes (active, ceased, and doubtful) into the hydroecoregions (HERs) that compose the Po River basin.

HER	**Active**	**Ceased**	**Doubtful**	**Tot.**
Cod. 1	3	8	3	**14**
Cod. 2	2	-	-	**2**
Cod. 3	-	9	1	**10**
Cod. 11	9	37	1	**47**
Cod. 18	24	25	3	**52**
Cod. 21	-	-	-	**-**
Cod. 24	2	3	-	**5**

Cod. 71	5	30	1	**36**
Cod. 132	293	1076	45	**1414**
Tot.	**338**	**1188**	**54**	**1580**

Table S3. Temporal evolution of number and area of pit lakes (PLs) in the subsample areas: TO (Turin), OR (Po and Orba River Park), MI (Milan), TR (Trezzo sull'Adda), BS (Brescia), MN (Mantua), MO (Modena) and PO (Po River shaft). Landsat-5 (L5), Landsat-7 (L7), Landsat-8 (L8) and Sentinel-2 (S2).

	Count *[n]*				**Area Tot.** *[ha]*			
	L5	**L7**	**L8**	**S2**	**L5**	**L7**	**L8**	**S2**
	1990	*2000*	*2014*	*2021*	*1990*	*2000*	*2014*	*2021*
TO	31	32	33	33	193	254	358	441
OR	11	15	25	25	70	95	142	215
MI	50	53	56	56	317	361	494	633
TR	10	10	13	13	33	41	67	91
BS	36	42	55	58	131	221	330	463
MN	19	21	24	24	53	53	66	123
MO	29	32	40	41	83	143	204	281
PO	14	30	66	70	28	107	518	735
Tot.	**200**	**235**	**312**	**320**	**909**	**1275**	**2179**	**2980**

Table S4. Maximum likelihood classification in pit lake buffer zones in the subsample areas.

	TO [%]	**OR** [%]	**MI** [%]	**TR** [%]	**BS** [%]	**MN** [%]	**PO** [%]	**MO** [%]
Water bodies	3.6	1.7	0.2	0.8	0.1	1.4	5.6	2.9
Riparian vegetation	22.5	31.1	31.2	37.9	22.7	18.3	25.9	44.3
Cultivated farmland	19.6	20.7	21.6	18.7	31.7	37.3	17.3	27.4
Bare soil	46.8	40.3	28.1	12.5	20.9	34.6	46.4	13.0
Urban structures	3.9	1.7	15.3	13.3	16.8	5.2	1.9	8.5
Sand	3.6	4.5	3.6	16.7	7.9	3.3	2.8	3.9

Table S5. Results of the power models ($V=\alpha A^{\beta}$) used for computing the lake volume (V) from the surface area (A) of the pit lakes in the Po River basin. WBs (water bodies).

Lake typology	**α**	**β**	**A**	**R^2**	**References**
Lake with A<1.25 km^2	0.0328	1.236	km^2	0.83	Delaney et al. (2022)
Lentic WBs associated with river networks	0.0071	1.118	ha	0.79	Izmailova and Koornenkova (2022)
Lakes with A<0.30 km^2 in the floodplain	0.0112	1.210	km^2	0.76	This study

Table S6. Mean (min-max) denitrification rates in PLs (pit lakes) and W&Os (wetlands and oxbows) in the subsample area PO. Data are from [A]Nizzoli et al. (2020) and [B]Racchetti et al. (2011).

Water bodies		**Total denitrification** (mg N m^{-2} d^{-1})
PLs [A]	Shallow waters sediments (littoral)	61 (36-98)
	Hypolimnetic sediments	18 (0.3-39)
W&Os [B]	Wetlands in river networks (healthy)	114 (12-634)
	Isolated wetlands (deteriorated)	16 (0.6-78)

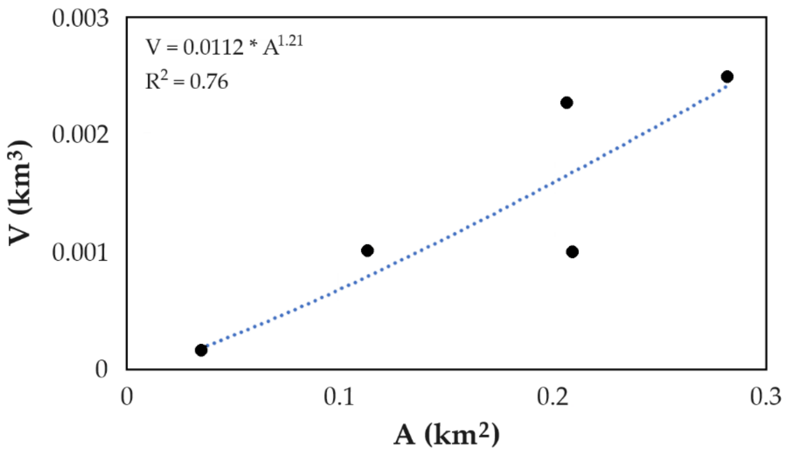

Figure S1. Relationship between surface (A, km^2) and volume (V, km^3) of five gravel and sand pit lakes in the subsample area PO. Original data are available in Nizzoli et al. (2020).

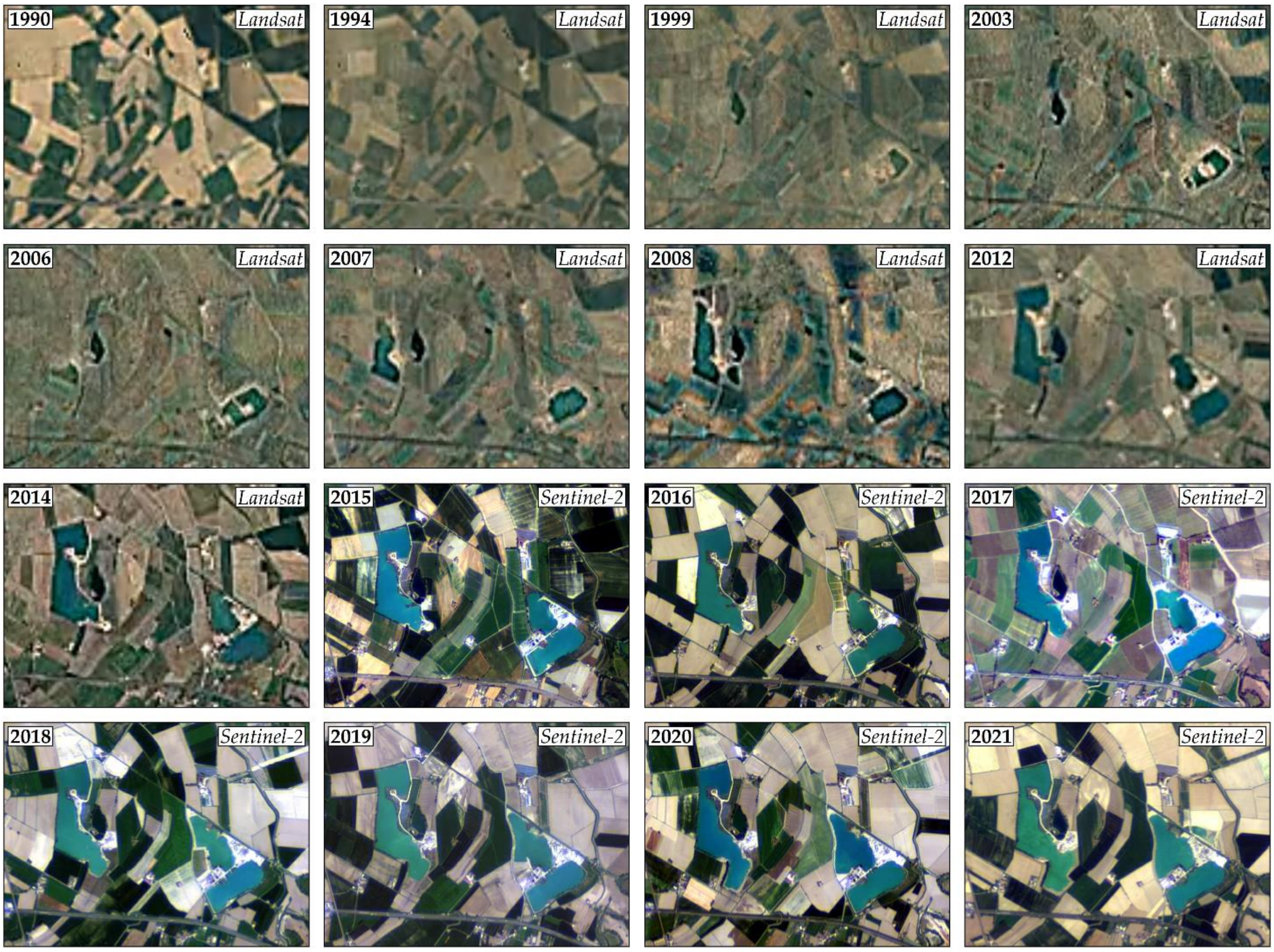

Figure S2. Temporal evolution of some pit lakes located in the PO area captured by multi-source satellite (Landsat-5, Landsat-7 and Sentinel-2). The spatial resolution is 30 m and 10 m for Landsat and Sentinel-2 imagery, respectively.

2.6 References

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Chapter III: The long-term detection of Suspended Particulate Matter concentration and water colour in gravel and sand pit lakes through Landsat and Sentinel-2 imagery

remote sensing

Open Access Article

The Long-Term Detection of Suspended Particulate Matter Concentration and Water Colour in Gravel and Sand Pit Lakes through Landsat and Sentinel-2 Imagery

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3.1 Introduction and aim of the study

Water colour is one of the oldest water observation parameters and is closely related to the optical properties of water, constituting a fundamental indicator for the optical water quality (Kirk, 1988). It is a result of water constituents and their interaction with sunlight, establishing the basis for water quality monitoring through optical remote sensing. This suggests that the information obtained from the colour of PLs can contribute to the assessment of their quality status. As a consequence, this parameter has also been recognised by the Global Climate Observing System as a fundamental climate variable for inland waters (Wang S. et al., 2021). SPM is another key parameter for describing water characteristics and can contribute to assessing aquatic ecosystems' quality (Pozdnyakov et al., 2005; Xing et al., 2013; Doxaran et al., 2014; Chen et al., 2016; Liu et al., 2017; Zhao et al., 2018; Ciancia et al., 2020; Jiang et al., 2021). It is a bio-optical parameter consisting of a mix of inorganic substances (e.g., mineral sediments), organic constituents (e.g., algal particles and vegetation debris) and water-insoluble microorganisms. In particular, most SPM manifest as complex, floc-like aggregate structures composed of a variety of minerals and organic matter ranging from the molecular to the organismal level (Walch et al., 2022). SPM concentration can directly and significantly influence the optical properties of water through the absorption and scattering of sunlight (Gernez et al., 2015; Hou et al., 2017). For example, it can directly reduce light penetration, affecting phytoplankton productivity and nutrient dynamics, as well as the living conditions of both aquatic animals and vegetation (Moore et al., 1997; Doxaran et al., 2002; Havens, 2003; Guildford et al., 2007; Zhu et al., 2013; Hou et al., 2017; Jiang et al., 2021; Ma et al., 2021; Wen et al., 2022). Therefore, monitoring temporal and spatial variations in SPM concentration is crucial for understanding the dynamics of aquatic ecosystems (Xing et al., 2013; Zhang et al., 2014; Gernez et al., 2015; Wu et al., 2015; Chen et al., 2016; Hou et al., 2017; Zhao et al., 2018; Lebeuf et al., 2019).

As a consequence, SPM plays a prominent role as an indicator to monitor the degradation of inland water resources and guide their management (Bilotta and Brazier, 2008; Williamson and Crawford, 2011; Shi et al., 2015). Particularly in inland waters, this parameter is often correlated with nutrient enrichment, such as that of nitrogen and phosphorus (Dekker et al., 2001; Hu et al., 2004), while in many high-turbid lakes, it has been directly associated with dredging activity (Fettweis et al., 2006). Given the importance of this parameter, many approaches and sensors have been adopted over the years to accurately estimate the SPM concentration from optical remote sensing data (Dekker et al., 2001; Doxaran et al., 2002; Miller and McKee, 2004; Han et al., 2006, 2016; Chen et al., 2007, 2015; Nechad et al., 2010; Feng et al., 2012, 2014; Dogliotti et al., 2015; Balasubramanian et al., 2020).
One of the advantages of Earth Observation (EO) data is its ability to obtain water quality information remotely over large areas and over the long term. It offers the opportunity to increase and improve the spatio-temporal coverage of inland water environmental monitoring (Lehmann et al., 2018). In recent years, remote sensing has become a low-cost operational tool that, in support of traditional limnological measurements, provides information on the state of surface waters by deriving bio-geophysical parameters, such as chlorophyll-a concentration (Matthews et al., 2012; Neil et al., 2019; Papenfus et al., 2020), turbidity (Potes et al., 2012; Güttler et al., 2013), suspended particulate matter (Song et al., 2014; Gao et al., 2021; Wen et al., 2022), phytoplankton types (Kutser, 2009), and Secchi disk depth (Jiang et al., 2019; Zhang et al., 2022). In particular, both water colour and SPM concentration products can be retrieved from satellite images.
Recently, an algorithm based on multispectral information acquired from satellite sensors has been proposed to derive the hue angle, an indicator that can be used to determine the λ_{dom} of a water body (i.e., the water colour) (Van der Woerd and Wernand, 2015, 2018). This indicator is called the Forel-Ule Index (FUI) and is derived from Remote Sensing Reflectances (Rrs). The FUI is not based on local retrieval algorithms; therefore, it can characterise natural waters easily and effectively (Ye and Sun, 2022; Wang S. et al. 2019). FUI, still used today, is a benchmark standard in numerous studies (Li et al., 2016; Wang, 2018; Wang et al., 2018, 2019, 2020, 2021; Giardino et al., 2019; Pitarch et al., 2019; Chen et al., 2020), and is characterised by having a relatively low uncertainty (Van der Woerd and Wernand, 2015; Van der Woerd and Wernand, 2018; Wang et al., 2018).
The monitoring of SPM concentration using traditional limnological techniques can provide accurate measurements; however, they are time-consuming, expensive and spatially limited (Leblanc et al., 2003; Puigserver et al., 2010; Di Polito et al., 2016; Liu et al., 2017; Cao et al., 2019). On the other hand, remote sensing techniques can be useful to complement *in situ* measurements, as they allow for large-scale, long-term observations of Visible-Near infrared (VIS-NIR) spectral regions which can be exploited to map SPM concentration (Jassby et al., 2003; Posch et al., 2012; Giardino et al., 2014; Zhang et al., 2014; Gernez et al., 2015; Shi et al., 2015; Wu et al., 2015; Dörnhöfer et al., 2016; Hou et al., 2017; Liu et al., 2017; Cao et al., 2019; Qin et al., 2019; Du et al., 2022).
In the present study, we assess the evolution of the λ_{dom} and the SPM concentration in a large sample of PLs and we expect to find a clear difference according to their sizes and locations, as well as in according with quarrying activity and seasonal variations. Our study seeks to examine the reliability of Landsat and Sentinel-2 satellites in estimating these two water quality parameters in small and dynamic aquatic environments such as PLs. The results highlight that location and size are the principal factors influencing water quality; moreover, it emerged that disturbance from quarrying activity does not have a long-term impact, because after the cessation of quarrying, SPM concentration decreases rapidly, although, on average, ceased PLs are characterised by higher λ_{dom} than those still active.

3.2 Materials and methods

3.2.1 *Study area*

The Po River basin extends around one of the largest rivers in the Mediterranean Sea and the longest river in Italy. The Po River is 652 km long and its basin covers approximately 74,000 km^2, of which ~71,000 km^2 are in Italy. In terms of water resources, the Po River (with an average annual flow of 1540 m^3 s^{-1} which has been gradually decreasing since the 21st century) is overexploited for irrigation, hydropower generation, and domestic purposes (Coppola et al., 2014; Montanari et al., 2023). The Po River basin represents a key territory for the economy of the entire country; in fact, economic activities within it account for about 40% of Italy's annual GDP (Viaroli et al., 2010; Musolino et al., 2017). In particular, the Po River basin contributes to about 60% of national sand and gravel production. The climax of this sector occurred from the post-World War II period until the 1980s, during which great amounts of inert materials were extracted directly from the riverbed to support post-war reconstruction, causing its gradual lowering (Lamberti, 1993). Once the severity of the river alteration was felt, mining activity was moved to the floodplain, leading to the creation of numerous PLs. For more details regarding water resources and quarrying activities in the Po River basin, see Ghirardi et al., (2023).

In the Po River basin, 1580 PLs have been identified, of which 338 were still active in 2021 (Ghirardi et al., 2023). These PLs differ in location (isolated, in proximity to or connected to a river), size (<1 to 52 ha), and quarrying activity (active or ceased). For more details regarding identification and classification of PLs, see Ghirardi et al., (2023).

For this study, a large subsample of PLs (320, both active and ceased) located in eight geographical areas within the basin was selected (Fig. 10): Turin (TO), Po and Orba River Park (OR), Milan (MI), Trezzo sull'Adda (TR), Brescia (BS), Mantua (MN), Modena (MO), and along the Po River shaft (PO). These areas were selected because they are spatially well-distributed and representative of different land uses (Ghirardi et al., 2023). In each of these geographic areas, the density of PLs is high and consequently they exhibit a high degree of heterogeneity and well represent the entire basin.

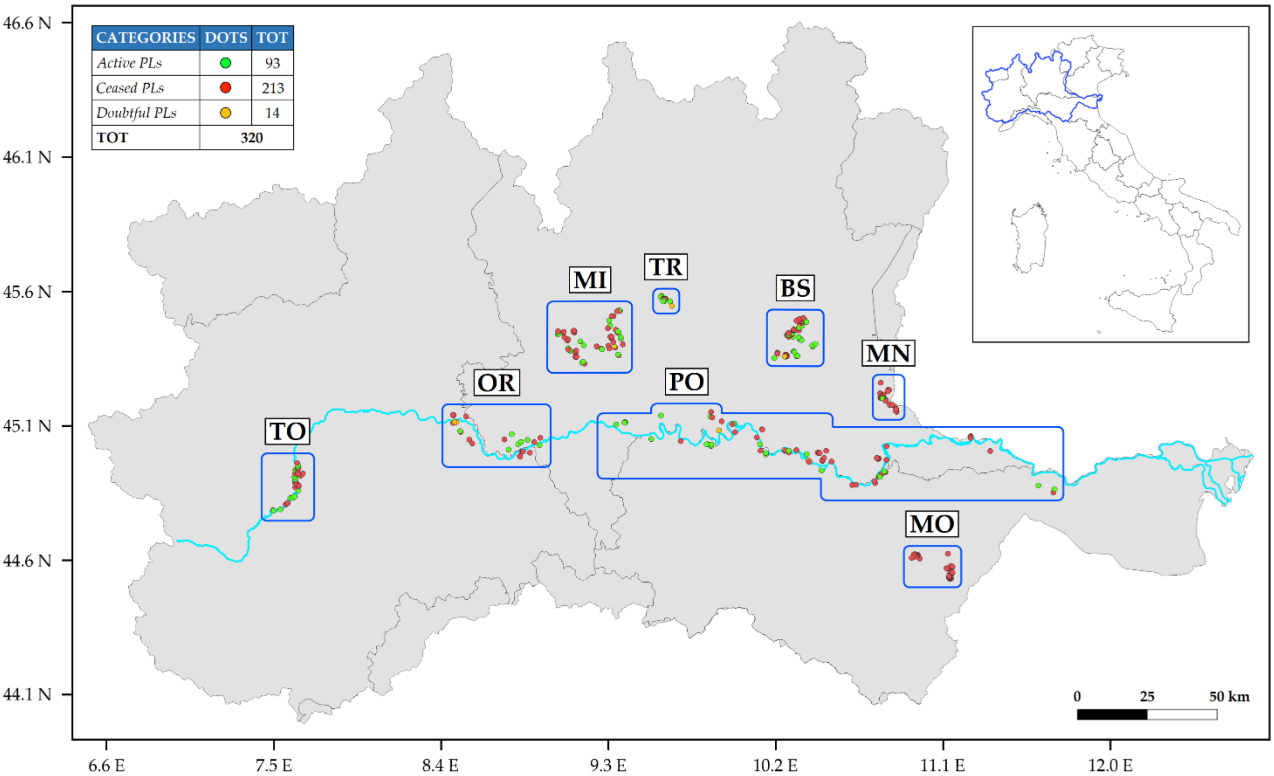

Figure 10. Pit lakes (PLs) divided into the eight subsample areas (blue boxes). The Po River is highlighted in light blue, green dots represent active PLs, red dots represent ceased PLs, and yellow dots represent doubtful PLs (all those are lakes that have the typical characteristics of PLs but whose origin or end of mining is uncertain). Turin (TO), Po and Orba River Park (OR), Milan (MI), Trezzo sull'Adda (TR), Brescia (BS), Mantua (MN), Modena (MO), and the Po River shaft (PO).

3.2.2 *The processing of satellite images*

Considering the wide time range used for this study (1990-2021), three different satellites were used: Landsat-5 (L5), Landsat-7 (L7) and Sentinel-2 (S2). The L5 and L7 belong to the Landsat constellation, and mount on board TM (Thematic Mapper) and ETM+ (Enhanced Thematic Mapper) sensors, respectively. They are characterised by a spatial resolution of 30 m, a revisit time of about 16 days and 5 bands in the VIS-SWIR (Visible-Short Wavelength InfraRed) domain. The S2 mission, on the other hand, comprises two polar satellites (S2A and S2B), placed on the same orbit, but offset 180° from each other, allowing a revisit time of about 5 days. They mount on board MultiSpectral Instrument (MSI) sensors, characterised by three different spatial resolutions (10, 20 and 60 m) and 13 bands in the VIS-SWIR.

All satellite images were downloaded as Level-1 (L1, i.e., not atmospherically corrected) from the following portals: https://catalogue.onda-dias.eu/catalogue/ (accessed on 29 November 2023) and https://earthexplorer.usgs.gov/ (accessed on 29 November 2023). All S2 images were resampled to the same spatial resolution (10 m). The years considered are listed in Table S7 and for each year, when available, six images were downloaded. Since there is no specific protocol for SPM concentration, we followed the protocol proposed by Buraschi et al., (2008) for sampling lake phytoplankton, according to the European Water Framework Directive (WFD): winter (January 1 – March 20), spring (April 1 – May 15), spring-summer (May 16 – June 15), summer (July 1 – August 31), summer-autumn (September 1 – October 1) and autumn (October 2 – November 31). A total of 375 satellite images divided into the eight subsample areas were downloaded, and these images were free of clouds and other radiometric problems (e.g., sunglint).

ACOLITE v.2022 (Vanhellemont and Ruddick, 2016, 2019) was used to atmospherically correct satellite images and to obtain SPM concentration maps. ACOLITE is a neural network that groups atmospheric correction algorithms and allows users to derive different water quality parameters from Rrs values. In addition, it is particularly suitable for the analysis of turbid and small inland water bodies. ACOLITE requires L1 satellite images as inputs and can mask all water pixels autonomously. The atmospheric correction algorithm used was Dark Spectrum Fitting (DSF), which is able to estimate Aerosol Optical Depth at 550 nm (AOD_{550}) from dark targets, while the algorithms used to estimate SPM concentration were: SPM_Nechad2010 (for Landsat images), adjusted with the *in situ* data specific to our case studies, and the SPM_Nechad2016 (for S2 images). The former was proposed by Nechad et al. (2010), while the latter was recalibrated in 2016 specifically for S2 images. Both exploit the spectral characteristics of the red band (630-690 nm). For more information regarding the two algorithms, see Nechad et al. (2010).

3.2.3 *Field campaigns and validation*

To validate SPM concentration maps obtained by processing Landsat images with ACOLITE code we used SPM concentration data collected by the University of Parma from 1993 to 2013. To this aim all cloud-free Landsat images with a maximum discrepancy of two days from *in situ* sampling were downloaded, totalling 76 images. To validate S2 data we carried out seven specific field campaigns (11 April 2022, 13 April 2022, 20 June 2022, 27 June 2022, 13 September 2022, 28 September 2022, and 15 June 2023) in PLs located in the PO area between 2022 and 2023 (Tab. S8). During these field campaigns, water samples were collected, and spectral signatures (Rrs) were acquired at the same sites. Water samples were filtered with Whatman GF/F fibre filters and used to determine gravimetrically the SPM concentration according to Strömbeck and Pierson (2001). Dry filters where subsequently incinerated in the muffle at 450 °C for 4 hours to obtain inorganic and organic fractions of the particulate. Reflectance measurements were collected using the handheld spectrometer WISP-3 (Hommersom et al., 2012) produced by Water Insight. The instrument is designed for water quality studies and can be used for the optical validation of satellite data. The optical range is from 400 to 800 nm, with a bandwidth (full width half maximum) of ~4.9 nm and is able to simultaneously measure water and sky radiances at 42° to the nadir (L_u and L_{sky}, respectively) and downwelling irradiance (E_d), using three different optics and appropriate geometry. Together, these three optics can be used to obtain the Rrs of the water surface. All measurements (5 replicates for every single station) were taken away from the shoreline to avoid any influence of the bottom on the radiometric measurements; in addition, measurements were taken at an azimuth angle of ~135° to the sun to avoid any sunglint effects. To compare the satellite data with *in situ* data, Regions of Interest (ROIs) (3x3 pixels) centred around the sampling site were created and the mean value was extracted for both SPM and Rrs; afterward, a series of descriptive statistics were calculated to assess their consistency. In detail, the determination coefficient (R^2), Mean Absolute Error (MAE), Root Mean Square Error (RMSE) and Mean Absolute Percentage Error (MAPE) were calculated for both parameters. In addition, for spectral signatures only, the Spectral Angle (SA) was also calculated, which was used to determine how similar the shape of satellite spectra is to *in situ* data (Braga et al., 2022; Pellegrino et al., 2023). The metrics used were computed as follows:

Eq. 6 $R^2 = 1 - \frac{\sum_{i=1}^{n}(x_i - y_i)}{\sum_{i=1}^{n}(x_i - \bar{x}_i)^2}$

Eq. 7 $MAE = \frac{\sum_{i=1}^{n}|x_i - y_i|}{n}$

Eq. 8 $\mathrm{RMSE} = \sqrt{\frac{\sum_{i=1}^{n}(x_i - y_i)^2}{n}}$

Eq. 9 $\mathrm{MAPE} = \mathrm{median}\left(\left|\frac{y_i - x_i}{x_i}\right| 100\%\right), i = 1, \ldots N$

Eq. 10 $\mathrm{SA} = \cos^{-1} \frac{\sum_{i=1}^{n} x_i y_i}{\sqrt{\sum_{i=1}^{n} y_i^2}\sqrt{\sum_{i=1}^{n} x_i^2}}$

Where x_i are the *in situ* values, $\bar{x}_i$ is the mean of the *in situ* values and y_i are the values derived from satellite images. In order to compare the spectral signatures acquired *in situ* (hyperspectral) with the spectral signatures of S2 (multispectral), a resampling of the hyperspectral signatures was performed based on spectral characteristics of S2. In particular, the higher spectral resolution of WISP data were spectrally resampled according to the full width half maximum (FWHM) of S2 data.

3.2.4 *Pit lakes analysis*

The identification of water pixels by ACOLITE is not always accurate because the pixels located at the water-land interface, which could lead to an overestimation of SPM concentrations, are often retained. For this reason, ACOLITE products were overlaid on the true colour images in order to verify the effective removal of terrestrial pixels and clouds. After that, starting from the ACOLITE water mask, all pixels located near the shorelines were manually removed. In addition, spectral signatures were examined for outliers within the water pixels. Once the ROIs were created, they were used to extract SPM concentration and Rrs values in the Visible domain, which were needed to obtain the λ_{dom} (water colour) via the Forel-Ule index (Van der Woerd and Wernand, 2018). More information on the procedure to obtain the λ_{dom} from Landsat and S2 images can be found in Van der Woerd and Wernand (2018). Finally, all outliers characterised by negative or excessively high values, often due to the presence of dredges in the middle of active PLs, were manually removed from these maps. Once the SPM concentrations were extracted from the PLs of all available images, the temporal trends of each lake and the average SPM concentration of each subsample area were calculated.

PLs were also divided into various thematic groups, with the purpose of understanding whether SPM concentration and λ_{dom} changed according to lake location, size, season (according to WFD protocol) and quarrying activity. Based on location, PLs were divided into three categories: isolated, in proximity to (at most 500 m away from a watercourse) or connected to a river. Based on dimension, they were divided into: small (<5 ha), medium (5-10 ha) and large (>10 ha). Based on season, the WFD protocol was followed, while based on quarrying activity, they were divided into: active and ceased. The Mann-Whitney and Kruskal-Wallis nonparametric tests were performed to assess whether there were significant differences in SPM and λ_{dom} among the categories that compose each thematic group.

To specifically assess the impact of quarrying activity on the evolution of the water quality of PLs, two types of analysis were conducted on a restricted number of PLs. The first involved the analysis of temporal trends of SPM concentration in 13 PLs whose quarrying activity ceased during the observed period (1990-2021). Specifically, the mean SPM concentrations before and after the end of quarrying activity was calculated for the 13 lakes, considering all available images. For this analysis, the cumulative precipitation during the seven days prior to the date of satellite acquisition was measured to understand the influence of this meteorological factor. The wind was not considered because the PLs examined were characterised by extensive areas of riparian vegetation in their surroundings (on average, 80% of the perimeter of those PLs were covered by vegetation). The second analysis concerned two adjacent PLs located in the MI area: one whose quarrying activity had ceased (MI-30), and the other which was characterised instead by frequent quarrying episodes (MI-31). For this analysis, 27 S2 images acquired during quarrying events were downloaded and processed with ACOLITE in order to obtain SPM concentration maps.

3.3 Results

3.3.1 *Satellite data validation*

The statistical comparison between the SPM concentrations obtained *in situ* and those derived from satellite images showed that there is a very strong correlation between both the data derived from Landsat images (R^2 = 0.85, MAE = 1.12, RMSE = 1.59, MAPE = 15.36%) and those from S2 images (R^2 = 0.82, MAE = 1.37, RMSE = 1.71, MAPE = 14.73%). However, it appeared that the SPM_Nechad2010 algorithm applied to the Landsat images overestimated the SPM concentration; consequently, two calibration coefficients were applied, one for the L5 and one for the L7 data, calculated on the basis of the slope of the regression lines. In both scatterplots, we removed a comparison related to a very turbid lake because, although there was a strong correlation between the *in situ* and satellite data, they distorted the R^2 values, over-improving them.
The comparison of the spectral signatures obtained using WISP-3 and those derived from ACOLITE showed that there is a strong correlation in the visible spectrum, while some issues emerge in the NIR domain (a slight overestimation of satellite values). This overestimation is not impactful for this work, since the algorithms for estimating the SPM concentration were based on the red band (630-690 nm), while the calculation of the λ_{dom} was based on the visible bands. Furthermore, the accuracy of the spectral signatures obtained through ACOLITE is also justified by the low SA values obtained for the 28 comparisons performed (10.8° ± 5.7°). The graphs for the statistical metrics are shown in Figures 11-12.

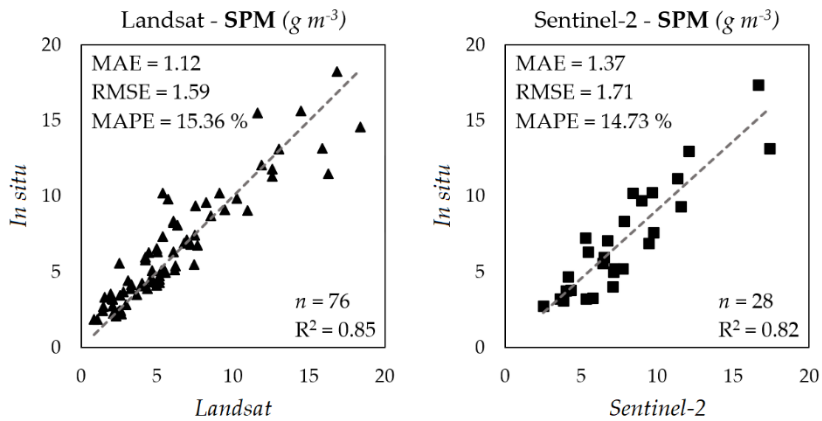

Figure 11. Scatterplots between *in situ* data (*y*-axis) and satellite data (*x*-axis). The black triangles and black squares represent comparisons between *in situ* and ACOLITE SPM concentrations from Landsat (L5 and L7, calibrated) and Sentinel-2 (S2) images, respectively. "n" represents the sample size, "R^2" the coefficient of determination, "MAE" the mean absolute error, "RMSE" the root mean square error, "MAPE" the mean absolute percentage error, and the dashed gray lines refers to the regression lines.

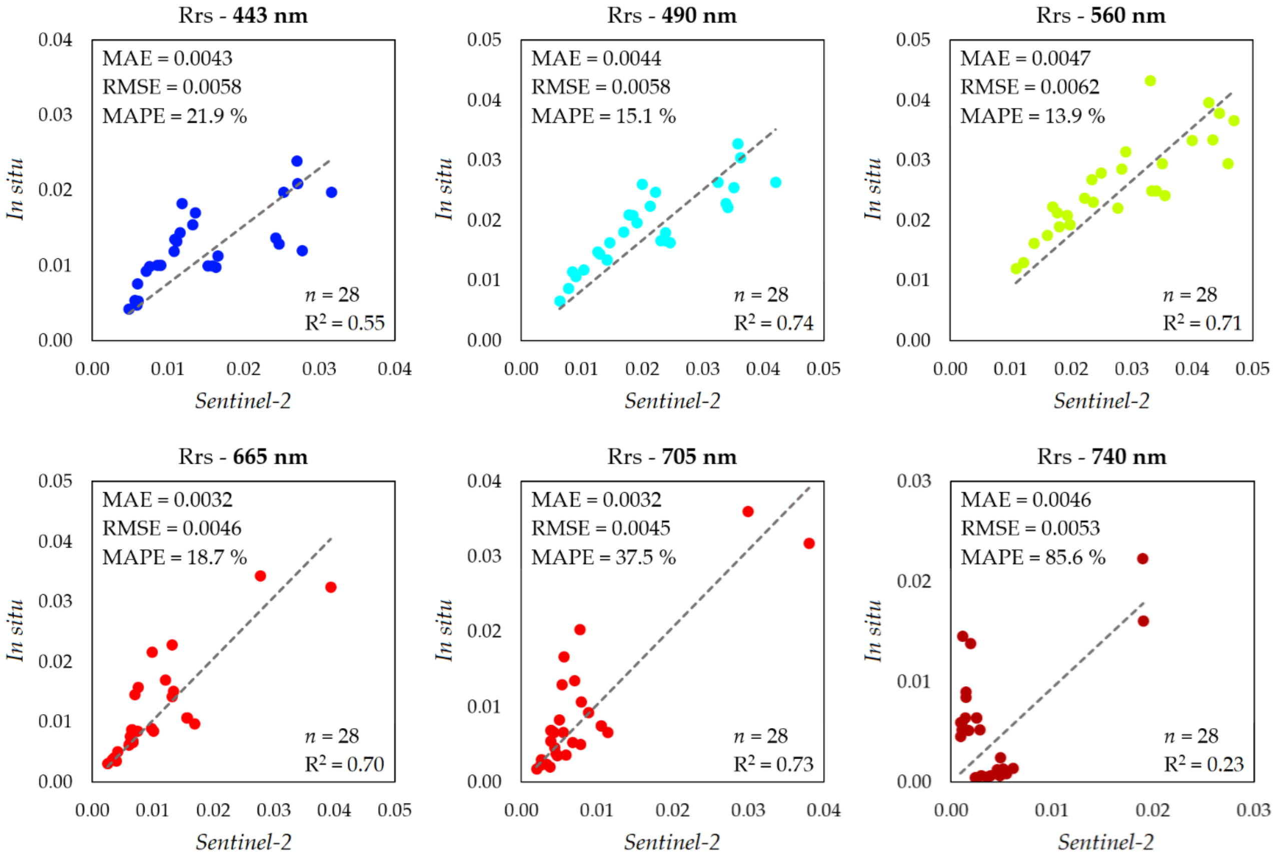

Figure 12. Scatterplots between *in situ* WISP-3 data (*y*-axis) and Sentinel-2 data (*x*-axis). The colored dots represent comparison between WISP-3 and ACOLITE spectral signatures, for the first six bands of S2. "n" represents the sample size, "R^2" the coefficient of determination, "MAE" the mean absolute error, "RMSE" the root mean square error, "MAPE" the mean absolute percentage error, and the dashed gray lines refers to the regression lines.

3.3.2 *SPM concentration and water colour*

Figure 13 shows the mean SPM concentration and the mean λ_{dom} (1990-2021) for all PLs of the subsample investigated. SPM values range from 1 to 88 gm^3, while λ_{dom} range from 480 to 589 nm. The PLs with the highest mean SPM concentrations are located along the Po River (especially in OR and PO areas) and more generally near rivers. For mean λ_{dom}, the pattern is also similar, although it is less pronounced than that of SPM concentration. In fact, PLs with high mean λ_{dom} are mainly located near rivers, although they are also present in other areas (e.g., MI area).

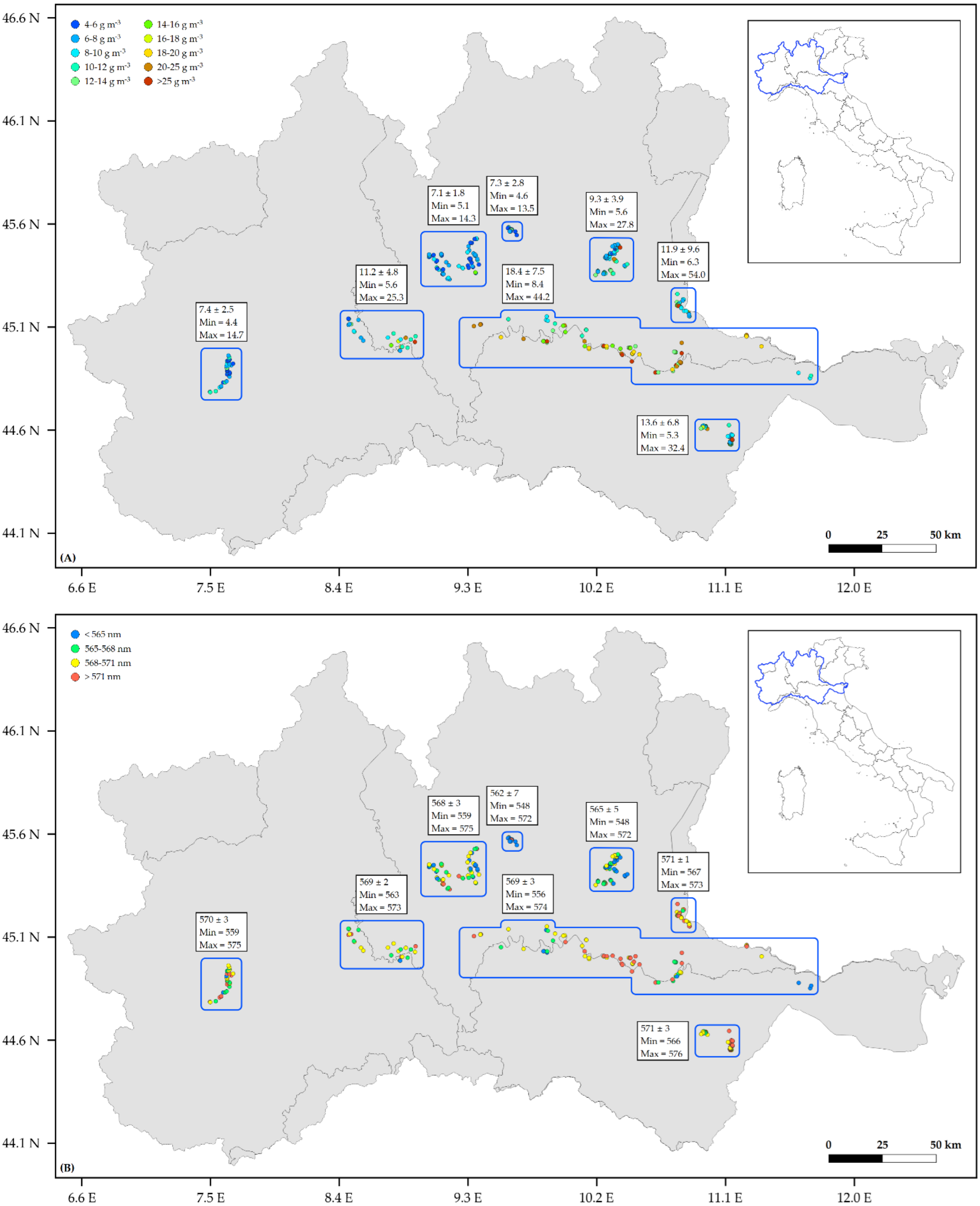

Figure 13. Mean SPM concentrations (A) and λdom (B) for subsample PLs for the period 1990-2021. The blue boxes represent the eight subsample areas. In the black boxes are the mean ± st.dev, and minimum and maximum SPM concentrations for each area.

Based on the PLs' locations, the analysis shows that those connected to a river (active: 21.4 ± 2.0 gm^{-3}; ceased: 15.7 ± 5.9 gm^{-3}) are characterised by higher mean SPM concentrations than both PLs located in proximity of a watercourse (active: 14.5 ± 7.9 gm^{-3}; ceased: 12.8 ± 7.4 gm^{-3}) and isolated ones (active: 12.4 ± 8.2 gm^{-3}; ceased: 8.9 ± 3.7 gm^{-3}), for both active and ceased PLs. The same result also emerges for the mean λ_{dom}. Based instead on size, smaller PLs (active: 18.2 ± 11.5 gm^{-3}; ceased: 11.4 ± 6.6 gm^{-3}) have the highest mean SPM concentrations compared to medium (active: 14.3 ± 7.2 gm^{-3}; ceased: 9.3 ± 4.2 gm^{-3}) and large lakes (active: 10.7 ± 5.3 gm^{-3}; ceased: 9.7 ± 5.1 gm^{-3}). In this case, the difference between the three categories is more pronounced for active PLs than for ceased ones. The mean λ_{dom} follows the same pattern only for ceased PLs (571 ± 3, 569 ± 3, 568 ± 3 nm, respectively), while smaller active PLs are characterised by lower mean λ_{dom} than medium sized PLs (565 ± 7, 568 ± 4, 565 ± 5 nm, respectively). Seasonal comparison shows the same trend for both SPM concentration and λ_{dom}: higher values in winter gradually decrease until spring-summer, then increase again. No differences are visible between active and ceased PLs. Finally, the comparison based on quarrying activity shows that active PLs (13.3 ± 8.2 gm^{-3}) exhibit higher mean SPM concentration than ceased PLs (10.6 ± 5.9 gm^{-3}), however, ceased PLs (570 ± 3 nm) show higher mean λ_{dom} than active PLs (566 ± 6 nm). All comparisons are shown in Figure 14, while the values divided into the eight subsample areas are reported in Table S9 and Table S10.

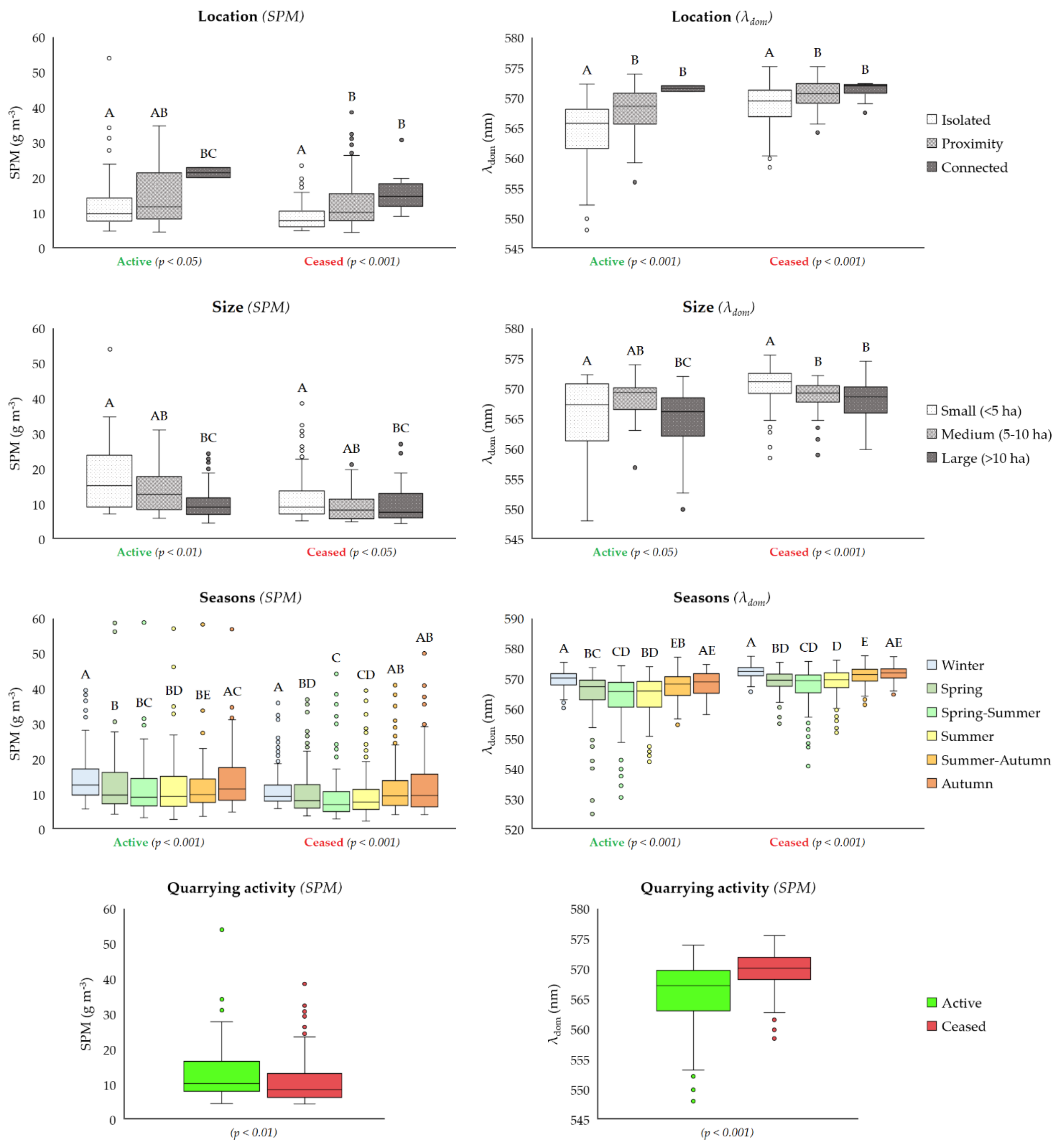

Figure 14. Boxplots of SPM concentrations (left column) and dominant wavelength (right column) of pit lakes (PLs) in the subsample (mean values referring to 1990-2021). Shown from top to bottom are comparisons of PLs' locations, PLs' sizes, season (according to WFD protocol), and quarrying activity. For location, size, and season boxplots, both active (left) and ceased (right) PLs are represented. In each boxplot, the circles represent the outliers. The "p" values refer to the Kruskal-Wallis and the Mann-Whitney (for quarrying boxplots only) statistical tests. For the Kruskal-Wallis test, the significant difference between two pairs of categories ($p < 0.05$) is indicated by the lack of identical letters.

By comparing the PLs characterised by low mean values of SPM and λ_{dom} (PLs $_{low}$) with the PLs characterised by high mean values of SPM and λ_{dom} (PLs $_{high}$), it was observed that the PLs $_{low}$ are larger (mean: 12 ha) and older (and consequently deeper). These PLs are predominantly isolated and consistently exhibit lower mean SPM concentrations compared to the overall mean of the subsample area where they are located. On the other hand, PLs $_{high}$ are smaller (mean: 5 ha) and younger (and consequently shallower). They are predominantly located in proximity or directly connected to a river, and consistently display higher mean SPM concentrations compared to the overall mean of the subsample area in which they are located.

3.3.3 *The impact of quarrying activity and precipitation*

Table 2 shows the mean SPM concentration of the 13 PLs before and after the cessation of quarrying activity (Fig. S3-S7). The results show that the mean SPM concentration decreases from a minimum of 43% to a maximum of 72% (mean reduction of 54%).

Table 2. SPM concentration (mean ± st.dev.) before and after the end of quarrying activity for the 13 pit lakes (PLs) examined (observation period: 1990-2021).

PLs	Coordinates	Quarrying activity	Before *($g\ m^{-3}$)*	After *($g\ m^{-3}$)*	SPM reduction *(%)*
OR-4	45.161294 N; 8.548939 E	1995-2012	9.7 ± 4.4	5.5 ± 3.6	-43
OR-22	45.031655 N; 8.873921 E	2005-2008	18.1 ± 7.7	7.4 ± 3.9	-59
OR-25	45.070040 N; 8.895304 E	2003-2007	19.8 ± 6.8	7.7 ± 3.2	-61
BS-11	45.378769 N; 10.181337 E	2006-2012	17.3 ± 7.8	8.5 ± 4.8	-51
BS-42	45.464501 N; 10.250156 E	<1990-2005	12.5 ± 5.7	5.8 ± 2.5	-54
BS-49	45.491103 N; 10.263158 E	<1990-2007	11.6 ± 6.0	6.6 ± 3.2	-43
MN-2	45.245806 N; 10.699478 E	1999-2007	14.3 ± 7.9	7.3 ± 2.8	-49
MO-5	44.673111 N; 10.817644 E	2006-2014	20.4 ± 6.9	5.9 ± 2.6	-71
PO-9	45.059638 N; 9.775130 E	<1990-2009	15.0 ± 8.1	7.3 ± 3.0	-51
PO-14	45.155322 N; 9.801703 E	2002-2012	19.4 ± 12.9	5.5 ± 1.9	-72
PO-17	45.141368 N; 9.849019 E	2002-2012	16.9 ± 7.2	8.3 ± 3.2	-51
PO-54	44.911351 N; 10.623380 E	1998-2012	20.4 ± 10.1	10.4 ± 5.4	-49
PO-77	44.861300 N; 11.524369 E	<1990-2008	14.0 ± 5.9	7.0 ± 3.3	-50

A clear correlation between SPM concentration and cumulative precipitation was not found for the 13 PLs analysed. For these comparisons, we exclusively considered values obtained after the ending of the quarrying activity. This approach ensured the elimination of any variability introduced by the resuspension associated with dredging.
Figure 15 shows the SPM concentration of PLs MI-30 (ceased) and MI-31 (active) during some quarrying events. The results show that MI-31 is consistently characterised by higher SPM concentrations than MI-30, especially in the southern part where the dredge is located. In addition, it is evident how the SPM peak located in the excavation area moved and extended rapidly within a few days to the entire lake (e.g., 16 June 2018 to 21 June 2018, 9 September 2019 to 14 September 2019, 10 June 2021 to 15 June 2021).

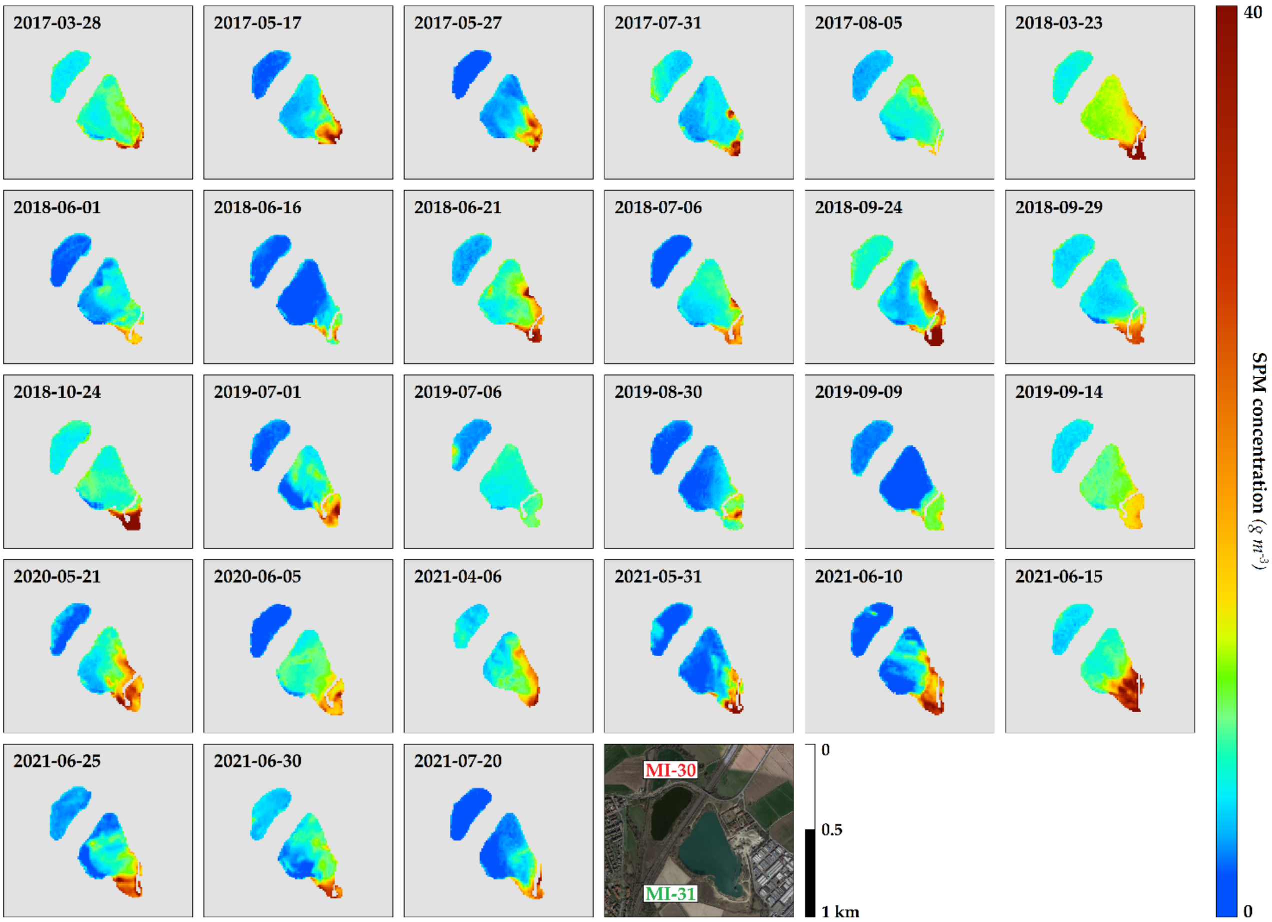

Figure 15. SPM concentration maps of pit lakes MI-30 (ceased) and MI-31 (active) from 2017 to 2021 referred to quarrying events. MI-30 (45.406015 N, 9.237314 E); MI-31 (45.403253 N, 9.241183 E). SPM concentration maps were obtained from Sentinel-2 Level 1 images, processed with ACOLITE (SPM_Nechad2016 algorithm).

3.4 Discussion

3.4.1 *The reliability of remote sensing for PLs' water quality assessment*

Satellites used for land monitoring, such as Landsat and Sentinel-2, provide images characterised by high spatial resolution that enable the assessment of water quality parameters. In particular, recent studies have corroborated the capabilities of such satellites in monitoring inland waters and small lakes (Toming et al., 2016; Liu et al., 2017; Ogilvie et al., 2018 a,b; Pahlevan et al., 2019; Ambrose-Igho et al., 2021). This success can be attributed to the increase in the average revisit interval, which has resulted in a concomitant increase in the number of cloud-free images. Such an abundance of data, therefore, allows a near real-time monitoring of such dynamic environments. In this study, we conducted an in-depth analysis of the reliability of Landsat-5, Landsat-7 and Sentinel-2 satellites in assessing the physical and optical properties of PLs focusing on the estimation of SPM concentration and λ_{dom}. Our approach included the comparison of SPM concentration data obtained *in situ* by limnological techniques with those retrieved through the use of the neural network ACOLITE on satellite data. The results of this comparison revealed a remarkable agreement; however, the SPM_Nechad2010 algorithm employed for the Landsat images showed a tendency to overestimate SPM concentrations. It was necessary to correct this overestimation by a recalibration process based on the reference data obtained *in situ*. The validation was mainly focused on PLs characterised by SPM concentrations below 20 gm^{-3}, due to a logistic constrain that did not allow the access to many PLs for field measurements. However, although only a few data were available for PLs with particularly high SPM concentrations, the agreement between *in situ* and satellite values was very strong. Nevertheless, these data were not included in the main validation step, as they would have distorted the scatterplots (Fig. 11), over-enhancing the coefficient of determination (R^2). In the future, it would be appropriate to find accessible PLs with SPM concentrations above 20 gm^{-3}. This would allow a more complete and detailed validation of the performance of the ACOLITE algorithm in environments characterised by such levels of SPM concentration. Moreover, it would be interesting to identify a few sample lakes and analyse their temporal evolution in detail through the integration of S2 data with products derived from, for example, Landsat-8 and Landsat-9 images, in order to avoid temporal gaps between data.

In addition to the SPM concentration, we also performed a comparison of the Rrs spectral signatures, obtained by the atmospheric correction of S2 images through ACOLITE, with those acquired through the Water Insight spectrometer WISP-3. This spectral parameter is essential for the calculation of λ_{dom}, and consequently, a robust agreement between the values obtained *in situ* and those detected by satellites is crucial. The comparison reveals strong agreement in the visible domain (400-700 nm), while some discrepancies were found in the NIR domain (a slight overestimation of satellite values). Specifically, the 740 nm band showed very good effectiveness for turbid waters as the presence of high SPM concentrations increases reflectance levels in both the red and NIR domains. However, this band is not as effective in clearer waters, where reflectance values should tend to zero. This phenomenon could result from an error in atmospheric correction due to the intrinsic properties of water in the infrared domain (De Keukelaere et al., 2018; Kuhn et al., 2019). This overestimation does not have a significant impact on our investigation, since the two algorithms used to estimate SPM concentrations (SPM_Nechad2010 and SPM_Nechad2016) were based on the red band (630-690 nm), while the calculation of λ_{dom} was based exclusively on the visible bands. More generally, the strong agreement between the spectral signatures collected *in situ* and those obtained from satellite data was confirmed by the low values obtained from the calculation of the SA statistical index (10.8° ± 5.7°).

3.4.2 *The assessment of PLs water quality*

The use of a combined approach between Landsat and Sentinel-2 satellite images allowed for the estimation of the mean SPM concentration and mean λ_{dom} for each PL over a three-decade period (1990-2021). Most of the challenges in inland water observations are due to their optical complexity. These aquatic systems can be a mixture of optically shallow and deep waters, with gradients of oligotrophic to hypertrophic productive waters and clear to turbid conditions. Hence, a large range in optical absorption and backscattering resulting in high optical variability can be found among and within lakes. This creates a challenge for algorithms applied to optical remote sensing for water-quality monitoring. Furthermore, another challenge is performing atmospheric corrections over such variable aquatic systems, as their complexity requires different approaches than those for land and ocean applications. The results of this temporal analysis revealed that the SPM values range from 1 to 88 gm^3, while λ_{dom} range from 480 to 589 nm. Regarding SPM concentration, there are regulations governing water use in agronomy and agriculture, which makes it even more important to understand the variations of this parameter over time. For example, the Food and Agriculture Organization of the United Nations (FAO) has suggested a maximum limit of 30 gm^{-3} for water use in agriculture. In Italy, on the other hand, Legislative Decree 185/2003 established a maximum limit of 10 gm^{-3} for wastewater reuse in agriculture.

The PLs with the highest mean SPM concentrations and highest mean λ_{dom} are located along the Po River (especially in the OR and PO areas) and, in general, near waterways and rivers. This finding is further confirmed through the subdivision of PLs into thematic groups; in fact, PLs connected to a river and those located in their proximity exhibited higher mean SPM concentrations and mean λ_{dom} than isolated PLs. These PLs are fed by river waters and, especially during flood periods, receive a large amount of suspended solids and nutrients that can lead to a higher trophic state (Nizzoli et al., 2020). An example of a river flood influencing the SPM concentration of a PL connected to a river is shown in Figure S8. Recent studies have shown that, as a consequence of climate change, periods of prolonged drought followed by short periods of flooding after intense precipitation are becoming more frequent and intense in the Po basin (Montanari et al., 2023). Thus, our results suggest that there is a significant potential for climate change to affect the water quality of PLs, particularly those connected to rivers, with prolonged periods of higher transparency during droughts and more turbid waters following flood events. Focusing on quarrying activity, it turned out, as might be expected, that active PLs are characterised by a higher mean SPM concentration than ceased PLs. This is mainly due to the mechanical action of the dredgers, which collect inert materials from the lake bottom and cause considerable resuspension of bottom sediments and stirring of suspended material. On the other hand, the mean λ_{dom} analysis shows that the ceased PLs show a higher mean λ_{dom} than the active PLs. Specifically, ceased PLs tend more toward a green-yellow colour, a feature attributable to the moderate presence of phytoplankton and Coloured Dissolved Organic Matter (CDOM). In fact, the absence of turbulence caused by dredges may contribute to greater water clarity, allowing sunlight to penetrate deeper and potentially promoting more algal particle growth. In addition, the high cover of riparian vegetation in the surroundings of the ceased PLs could result in an increased concentration of CDOM, via inputs of allochthonous dissolved organic matter (Jonsson et al., 2001; Pagano et al., 2014).

The division of PLs according to size revealed that small lakes (<5 ha) on average have higher SPM concentrations than their larger counterparts. This trend is more pronounced in PLs that are still active, while it tends to diminish in those that have ceased the quarrying activity. The explanation for this phenomenon may lie in the fact that, in smaller active PLs, quarrying activity has only recently started. As a result, such lakes are generally shallow and subject to the mechanical action of dredges, and to wind and fish action. These phenomena can resuspend lake sediment from the bottom, causing a drastic increase in SPM concentrations. The same trend is also reflected in the analysis of the mean λ_{dom} in ceased PLs, while for active PLs, the results show that small-sized PLs are characterised by a lower λ_{dom} than medium-sized PLs. This result could be attributed to the sparseness or absence of riparian vegetation in small-sized active PLs. In fact, the presence of riparian vegetation may lead to the introduction of organic debris into the waters, which, once dissolved, may contribute to increases in the λ_{dom}.

Finally, the increase in SPM concentration during the winter and autumn periods can be attributed to water mixing, increased wind and rainfall, and a reduction in riparian vegetation; factors that promote the input of particulate material from outside. In addition, more organic debris may flow in lake waters during these periods, leading to an increase in CDOM. On the other hand, the green hue of the waters, predominant in the spring and summer months, may be explained by algal blooms, caused by increased nutrients reaching the lake from agricultural fields, associated with higher temperatures.

3.4.3 *The impact of quarrying activity and precipitation*

Quarrying activity has been proven to have a considerable impact on PLs. For example, the turbulence generated within the water column disrupts thermal stratification and resuspends significant amounts of sediment, exerting a significant influence on the lake's planktonic

community (Agbeti et al. 1997; Alvarez-Cobelas et al. 2006; Burford and O'Donohue 2006). At the same time, sediment resuspension contributes to increased turbidity in the water, resulting in decreased sunlight penetration. This phenomenon, in addition to reducing the amount of light available to phytoplankton, influences macrophyte communities that may establish along banks characterised by gentle slopes. Such communities play a crucial role in maintaining high water quality standards (Wang D. et al., 2022), as well as providing habitats and food sources for numerous aquatic organisms (Mollema and Antonellini, 2016; Søndergaard et al., 2018).
The characteristics of the MSI sensor installed on board the S2 have been proven suitable for analysing the impact of quarrying activity on the SPM concentration of PLs. This suitability extends to both the long-term, allowing for the evaluation of temporal trends in SPM concentrations before and after the cessation of quarrying activities, and the short-term, allowing for the detection of recurring and episodic turbidity events in specific PLs that occur at small spatio-temporal scales. However, despite a high spatial resolution and a short revisit time, the extent of some small, newly formed PLs and the presence of clouds at specific periods of the year may limit the observation of short, punctual quarrying episodes.
As was to be expected, the end of quarrying activity led to a marked decrease in SPM concentrations in all of the PLs examined (a mean reduction of 54%), mainly due to the cessation of the mechanical sediment-stirring action carried out by the dredges. However, even once quarrying activity was over, no direct correlation emerged between SPM concentrations and total precipitation, as might have been expected. This could depend on multiple factors, including the actual location of the weather station relative to the lake and the time scale adopted, as soil erosion responds not only to the total amount of rainfall but also to rainfall intensity. In our perspective, it would be interesting to identify a weather station near a lake and consider a change in the time interval considered for the total precipitation. A plausible hypothesis is that within the total precipitation, rainfall that occurs shortly after the satellite image acquisition may have a greater influence than rainfall that occurs days later. Moreover, there is a complex relation between precipitation intensity, plant biomass and composition and erosion (Nearing et al., 2005) that could influence water quality. This analysis is beyond the scope of this work but, given the observed changes in land cover due to climate change and anthropogenic activities observed in the Po river watershed (Ghaderpour et al., 2023), the approach adopted in this work could be useful in the future for understanding the consequences of climate change and directing anthropic activity on the water quality of PLs.
An additional finding from this research indicates that the resuspension action actuated by the dredges is not confined to a single well-defined point. On the contrary, the increase in the SPM concentration found at the sampling point spread gradually throughout the entire lake over the course of a few days (potentially even more rapidly, given the frequency of the revisiting of S2, which occurred approximately every 5 days).

3.5 Supplementary materials

Table S7. Landsat-5 (L5, red), Landsat-7 (L7, blue) and Sentinel-2 (S2, green) satellite images processed for subsample pit lake analysis. Turin (TO), Po and Orba River Park (OR), Milan (MI), Trezzo sull'Adda (TR), Brescia (BS), Mantua (MN), Modena (MO), and along the Po River shaft (PO). The seasons follow the WFD protocol: winter (Win.; January 1 - March 20), spring (Spr.; April 1 - May 15), spring-summer (Spr. – Sum.; May 16 - June 15), summer (Sum.; July 1 - August 31), summer-autumn (Sum. – Aut.; September 1 - October 1) and autumn (Aut.; October 2 - November 31).

Year	Season	TO	OR	MI	TR	BS	MN	MO	PO*
1990	*Win.*	23/02	23/02	-	-	-	-	-	-
	Spr.	12/04	12/04	12/04	12/04	07/05	07/05	07/05	07/05 + 30/04
	Spr. – Sum.	15/06	15/06	-	-	-	-	-	-
	Sum.	02/08	02/08	-	-	-	-	-	-
	Sum. – Aut.	10/09	-	-	-	-	-	-	-
	Aut.	05/10	06/11	-	-	-	-	-	-
1994	*Win.*	-	-	-	-	-	-	19/01	11/02 + 04/02
	Spr.	30/04	07/04	07/04	07/04	02/05	02/05	02/05	02/05 + NA
	Spr. – Sum.	-	-	-	-	-	-	27/05	19/06 + 27/05
	Sum.	-	-	-	-	-	-	30/07	06/08 + 30/07
	Sum. – Aut.	-	-	-	-	-	-	07/09	07/09 + NA
	Aut.	-	-	-	-	-	-	25/10	-
1999	*Win.*	-	31/01	15/01	15/01	24/01	24/01	-	25/02 + NA
	Spr.	05/04	05/04	05/04	05/04	29/03	-	30/04	30/04 + NA
	Spr. – Sum.	-	23/05	23/05	23/05	01/06	01/06	-	17/06 + 10/06
	Sum.	-	26/07	26/07	26/07	04/08	04/08	-	03/07 + 05/08
	Sum. – Aut.	-	12/09	12/09	12/09	13/09	14/09	-	13/09 + 14/09
	Aut.	-	23/11	23/11	23/11	24/11	07/10	-	07/10 + 08/10
2000	*Win.*	26/01	26/01	-	-	-	-	-	-
	Spr.	01/05	01/05	01/05	01/05	08/04	08/04	08/04	08/04 + 03/05
	Spr. – Sum.	09/06	02/06	-	-	-	-	-	-
	Sum.	28/08	21/08	-	-	-	-	-	-
	Sum. – Aut.	22/09	22/09	-	-	-	-	-	-
	Aut.	08/10	08/10	-	-	-	-	-	-
2003	*Win.*	-	-	-	-	-	-	21/02	28/02 + 21/02
	Spr.	24/04	24/04	24/04	24/04	17/04	17/04	01/04	01/04 + NA
	Spr. – Sum.	-	-	-	-	-	-	28/05	19/05 + 28/05
	Sum.	-	-	-	-	-	-	08/08	30/07 + 08/08
	Sum. – Aut.	-	-	-	-	-	-	16/09	16/09 + 25/09
	Aut.	-	-	-	-	-	-	11/10	-
2006	*Win.*	10/02	10/01	-	-	-	-	-	-
	Spr.	-	08/04	08/04	08/04	11/05	11/05	03/05	03/05 + NA
	Spr. – Sum.	10/06	11/06	-	-	-	-	-	-
	Sum.	30/08	30/08	-	-	-	-	-	-
	Sum. – Aut.	-	-	-	-	-	-	-	-
	Aut.	02/11	02/11	-	-	-	-	-	-
2007	*Win.*	-	05/01	15/02	30/01	30/01	06/01	-	-
	Spr.	27/04	27/04	20/04	20/04	20/04	20/04	20/04	20/04 + 29/04
	Spr. – Sum.	-	-	22/05	30/05	30/05	22/05	-	-
	Sum.	-	16/07	16/07	16/07	25/07	25/07	-	-

	Sum. – Aut.	-	18/09	02/09	02/09	11/09	12/09	-	-
	Aut.	-	29/11	21/10	29/11	-	21/10	-	-
2008	*Win.*	-	-	-	-	-	-	10/02	10/02 + 19/02
	Spr.	04/04	05/04	08/05	07/05	-	08/05	08/05	08/05 + 09/05
	Spr. – Sum.	-	-	-	-	-	-	10/06	09/06 + 10/06
	Sum.	-	-	-	-	-	-	20/07	11/07 + 04/07
	Sum. – Aut.	-	-	-	-	-	-	-	-
	Aut.	-	-	-	-	-	-	-	-
2012	*Win.*	11/01	11/01	11/01	11/01	21/02	21/02	-	21/02 + 13/01
	Spr.	31/03	-	31/03	31/03	25/04	25/04	04/05	11/05 + NA
	Spr. – Sum.	25/05	19/06	19/06	19/06	-	05/06	-	27/05 + 21/06
	Sum.	22/08	22/08	22/08	22/08	15/08	30/07	-	30/07 + 08/08
	Sum. – Aut.	14/09	07/09	07/09	07/09	16/09	09/09	-	16/09 + 09/09
	Aut.	25/10	25/10	09/10	09/10	02/10	02/10	-	02/10 + NA
2014	*Win.*	-	-	-	-	-	-	25/01	25/01 + NA
	Spr.	28/03	-	06/04	06/04	15/04	01/05	01/05	01/05 + 24/04
	Spr. – Sum.	-	-	-	-	-	-	11/06	17/05 + 11/06
	Sum.	-	-	-	-	-	-	05/08	05/08 + 14/08
	Sum. – Aut.	-	-	-	-	-	-	06/09	06/09 + 15/09
	Aut.	-	-	-	-	-	-	24/10	24/10 + 02/11
2015	*Win.*	-	-	-	-	-	-	-	-
	Spr.	-	-	-	-	-	-	-	-
	Spr. – Sum.	-	-	-	-	-	-	-	-
	Sum.	-	06/08	06/08	06/08	06/08	06/08	-	06/08 + 13/08
	Sum. – Aut.	-	25/09	25/09	25/09	25/09	25/09	-	25/09 + 12/09
	Aut.	-	24/11	24/11	24/11	24/11	24/11	-	24/11 + 22/10
2016	*Win.*	-	-	-	-	-	-	-	-
	Spr.	12/04	12/04	22/04	22/04	22/04	22/04	29/04	22/04 + 29/04
	Spr. – Sum.	-	-	-	-	-	-	-	-
	Sum.	-	-	-	-	-	-	-	-
	Sum. – Aut.	-	-	-	-	-	-	-	-
	Aut.	-	-	-	-	-	-	-	-
2017	*Win.*	07/01	-	-	-	-	-	24/01	16/02 + 14/01
	Spr.	07/04	07/04	17/04	17/04	17/04	17/04	14/05	17/04 + 14/05
	Spr. – Sum.	17/05	-	-	-	-	-	03/06	27/05 + 03/06
	Sum.	06/07	-	-	-	-	-	08/07	06/07 + 08/07
	Sum. – Aut.	12/09	-	-	-	-	-	26/09	NA + 26/09
	Aut.	14/10	-	-	-	-	-	16/10	18/11 + 16/10
2018	*Win.*	11/02	02/01	22/01	22/01	24/01	19/01	-	22/01 + 14/01
	Spr.	17/04	17/04	22/04	22/04	22/04	22/04	24/04	22/04 + 19/04
	Spr. – Sum.	01/06	01/06	01/06	01/06	24/05	19/05	-	01/06 + 03/06
	Sum.	26/07	26/07	31/07	31/07	31/07	31/07	-	31/07 + 28/07
	Sum. – Aut.	19/09	19/09	24/09	24/09	24/09	24/09	-	04/09 + 06/09
	Aut.	24/10	24/10	24/10	24/10	19/10	24/10	-	24/10 + NA
2019	*Win.*	-	-	-	-	-	-	-	-
	Spr.	15/04	17/04	17/04	17/04	17/04	17/04	19/04	17/04 + 19/04
	Spr. – Sum.	-	-	-	-	-	-	-	-
	Sum.	-	-	-	-	-	-	-	-

	Sum. – Aut.	-	-	-	-	-	-	-	-
	Aut.	-	-	-	-	-	-	-	-
2020	*Win.*	12/01	12/01	12/01	12/01	12/01	12/01	29/01	12/01 + 29/01
	Spr.	06/04	11/04	11/04	11/04	11/04	11/04	08/04	11/04 + 08/04
	Spr. – Sum.	26/05	21/05	21/05	21/05	02/06	23/05	23/05	21/05 + 23/05
	Sum.	25/07	25/07	25/07	25/07	25/07	25/07	27/07	25/07 + 27/07
	Sum. – Aut.	13/09	13/09	13/09	13/09	13/09	13/09	15/09	13/09 + 15/09
	Aut.	13/10	08/10	28/10	28/10	28/10	28/10	10/10	28/10 + 10/10
2021	*Win.*	26/01	16/01	16/01	16/01	25/02	16/01	13/01	16/01 + 13/01
	Spr.	06/04	06/04	06/04	16/04	08/04	08/04	08/04	21/04 + 08/04
	Spr. – Sum.	19/05	10/06	10/06	31/05	28/05	28/05	18/05	15/06 + 18/05
	Sum.	10/07	10/07	10/07	10/07	22/07	10/07	22/07	10/07 + 07/07
	Sum. – Aut.	13/09	13/09	13/09	13/09	13/09	13/09	20/09	13/09 + 20/09
	Aut.	18/10	18/10	18/10	18/10	18/10	18/10	15/10	18/10 + 15/10

* Two different images are needed for the PO area because of its size.

Table S8. *In situ* measurement campaigns to validate remote sensing (RS) products from Sentinel-2 satellite images.

Pit lakes	Sampling	Coordinate N	Coordinate E	Date – *in situ*	Date – *RS*
Laghi di Cavallara 1	I	44.99801	10.63943	11/04/2022	11/04/2022
Laghi di Cavallara 2	I	44.99971	10.63958	11/04/2022	11/04/2022
Laghi di Cavallara 3	I	44.99921	10.64321	11/04/2022	11/04/2022
Laghi di Cavallara 4	I	44.99815	10.64325	11/04/2022	11/04/2022
Cava Baita	I	44.93918	10.64777	11/04/2022	11/04/2022
	II	44.93559	10.64306	11/04/2022	11/04/2022
Polo Belgrado Fogarino 1	I	44.94800	10.67448	11/04/2022	11/04/2022
Polo Belgrado Fogarino 2	I	44.95198	10.66959	13/04/2022	13/04/2022
	II	44.95303	10.67153	13/04/2022	13/04/2022
	III	44.95168	10.67151	13/04/2022	13/04/2022
Isola degli Internati	I	44.91544	10.61520	13/04/2022	13/04/2022
Cava Malaspina	I	44.91118	10.62320	13/04/2022	13/04/2022
	II	44.91263	10.62333	13/09/2022	13/09/2022
Ca' Morta	I	45.06416	9.77108	20/06/2022	20/06/2022
Lago Verde	I	45.06013	9.77499	20/06/2022	20/06/2022
Ca' Stanga	I	45.05852	9.79616	20/06/2022	20/06/2022
Bella Venezia	I	45.05166	10.03599	28/09/2022	23/09/2022
	II	45.05327	10.03595	28/09/2022	23/09/2022

	III	45.05080	10.03601	15/06/2023	15/06/2023
	IV	45.05248	10.03609	15/06/2023	15/06/2023
	V	45.05457	10.03616	15/06/2023	15/06/2023
Polesine Sud	I	45.02177	10.06836	28/09/2022	23/09/2022
	II	45.02237	10.06633	28/09/2022	23/09/2022
	III	45.02200	10.06744	15/06/2023	15/06/2023
	IV	45.02269	10.06664	15/06/2023	15/06/2023
Polesine Nord	I	45.02463	10.06557	28/09/2022	23/09/2022
	II	45.02523	10.06797	15/06/2023	15/06/2023
	III	45.02463	10.06671	15/06/2023	15/06/2023

Table S9. Mean SPM concentrations (gm^{-3}) divided into the four categories (location, dimension, season and quarrying activity) and into the eight subsample areas: Turin (TO), Po and Orba River Park (OR), Milan (MI), Trezzo sull'Adda (TR), Brescia (BS), Mantua (MN), Modena (MO), and along the Po River shaft (PO). Small (S.), medium (M.) and Large (L.). Winter (Win.; January 1 - March 20), spring (Spr.; April 1 - May 15), spring-summer (Spr. – Sum.; May 16 - June 15), summer (Sum.; July 1 - August 31), summer-autumn (Sum. – Aut.; September 1 - October 1) and autumn (Aut.; October 2 - November 31).

		TO	**OR**	**MI**	**TR**	**BS**	**MN**	**MO**	**PO**	**Active**	**Ceased**	**TOT**
Location	*Isolated*	6.4	9.7	7.0	7.4	9.3	14.1	9.1	16.8	12.4	8.9	10.0
	Proximity	7.7	13.4	8.2	7.1	-	8.7	14.6	18.7	14.5	12.8	13.2
	Connected	5.5	-	-	-	-	6.5	-	18.8	21.4	15.7	16.5
Dimension	*Small (<5 ha)*	7.4	9.3	7.3	8.2	9.2	13.6	14.4	20.1	18.2	11.4	12.6
	Medium (5-10 ha)	7.0	14.8	6.9	-	11.0	9.5	12.1	15.4	14.3	9.3	10.8
	Large (>10 ha)	7.6	9.6	7.1	5.2	8.3	6.7	11.9	17.0	10.7	9.7	10.1
Season	*Winter*	9.4	12.1	9.4	8.0	11.9	11.1	12.3	18.5	15.1	11.4	12.4
	Spring	6.6	12.8	6.8	7.8	8.3	12.0	12.8	18.4	12.8	10.5	11.2
	Spring – Summer	7.5	10.1	5.8	6.7	7.3	10.9	12.9	14.3	11.6	9.0	9.7
	Summer	6.6	7.8	6.1	5.7	9.1	10.9	12.4	17.1	12.2	9.6	10.4
	Summer – Autumn	7.5	11.6	7.7	6.6	9.0	13.4	16.2	17.1	11.7	11.8	11.7
	Autumn	7.6	12.4	7.0	8.3	10.2	13.2	16.5	20.5	14.4	12.1	12.7
Quarrying activity	*Active*	8.9	15.0	8.3	6.7	12.0	-	-	20.4			13.3
	Ceased	6.4	9.7	6.4	8.4	7.5	9.7	13.6	16.6			10.6

Table S10. Mean λ_{dom} (nm) divided into the four categories (location, dimension, season and quarrying activity) and into the eight subsample areas: Turin (TO), Po and Orba River Park (OR), Milan (MI), Trezzo sull'Adda (TR), Brescia (BS), Mantua (MN), Modena (MO), and along the Po River shaft (PO). Small (S.), medium (M.) and Large (L.). Winter (Win.; January 1 - March 20), spring (Spr.; April 1 - May 15), spring-summer (Spr. – Sum.; May 16 - June 15), summer (Sum.; July 1 - August 31), summer-autumn (Sum. – Aut.; September 1 - October 1) and autumn (Aut.; October 2 - November 31).

		TO	**OR**	**MI**	**TR**	**BS**	**MN**	**MO**	**PO**	**Active**	**Ceased**	**TOT**
Location	*Isolated*	571	569	568	562	565	571	573	569	564	569	567
	Proximity	570	570	567	563	-	571	571	570	568	571	570
	Connected	570	-	-	-	-	572	-	571	572	571	571
Dimension	*Small (<5 ha)*	573	570	571	563	566	571	572	570	565	571	570
	Medium (5-10 ha)	570	569	569	-	566	570	570	570	568	569	569
	Large (>10 ha)	567	569	566	561	560	571	569	568	565	568	566
Season	*Winter*	573	572	571	568	569	572	572	571	569	572	571
	Spring	569	568	567	558	563	570	570	568	564	569	567
	Spring – Summer	569	567	565	555	557	570	570	569	562	567	565
	Summer	568	568	565	560	564	570	572	568	564	569	567
	Summer – Autumn	570	571	569	566	566	571	573	570	567	571	569
	Autumn	571	571	571	567	568	573	572	571	568	571	570
Quarrying activity	*Active*	567	569	566	559	562	-	-	568			566
	Ceased	572	569	569	566	567	571	571	570			570

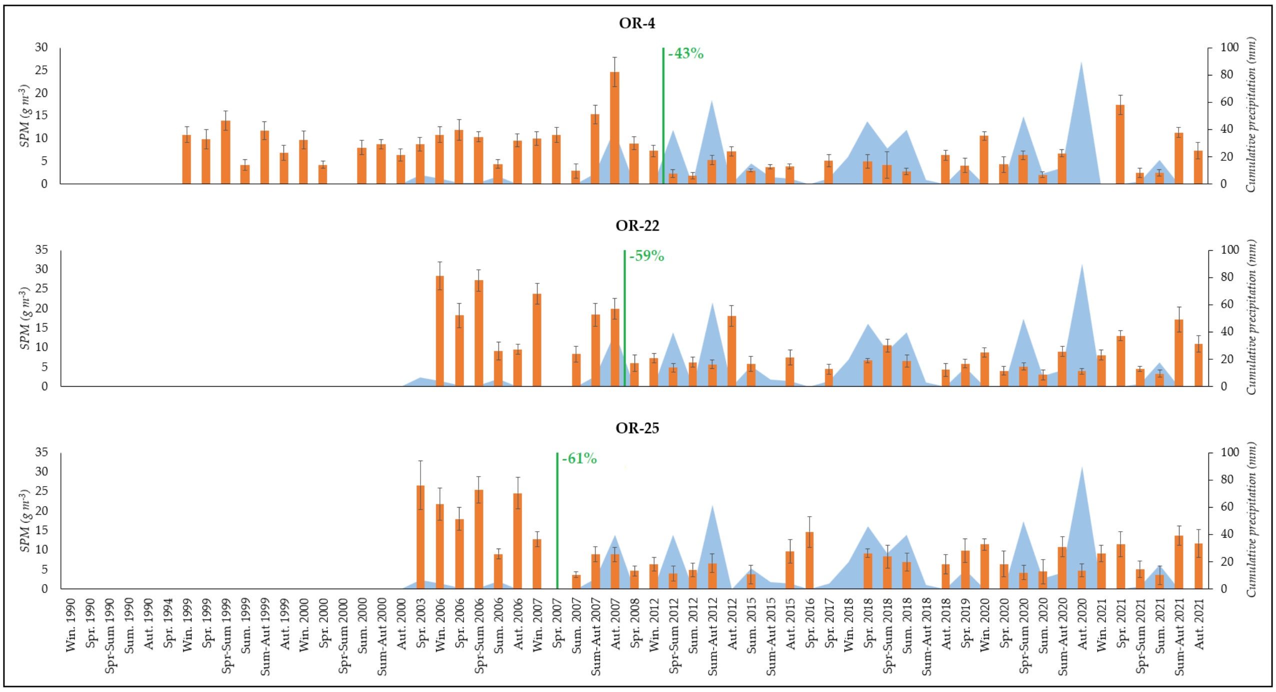

Figure S3. Temporal evolution of SPM concentration (orange columns) in some pit lakes in the OR area (Po and Orba River Park) in relation to cumulative precipitation (blue polygons) in the 7 days prior to satellite acquisitions. The green line indicates the end of quarrying activities and the mean percentage decrease in SPM concentrations after that event.

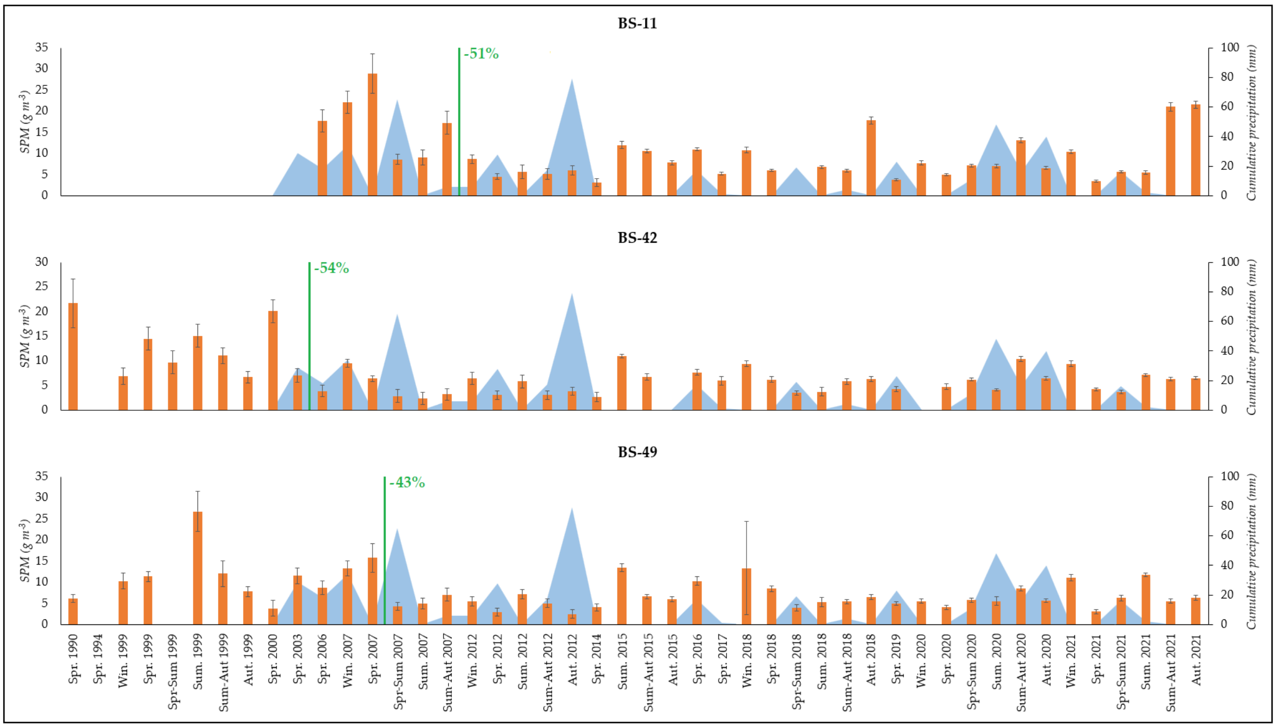

Figure S4. Temporal evolution of SPM concentration (orange columns) in some pit lakes in the BS area (Brescia) in relation to cumulative precipitation (blue polygons) in the 7 days prior to satellite acquisitions. The green line indicates the end of quarrying activities and the mean percentage decrease in SPM concentrations after that event.

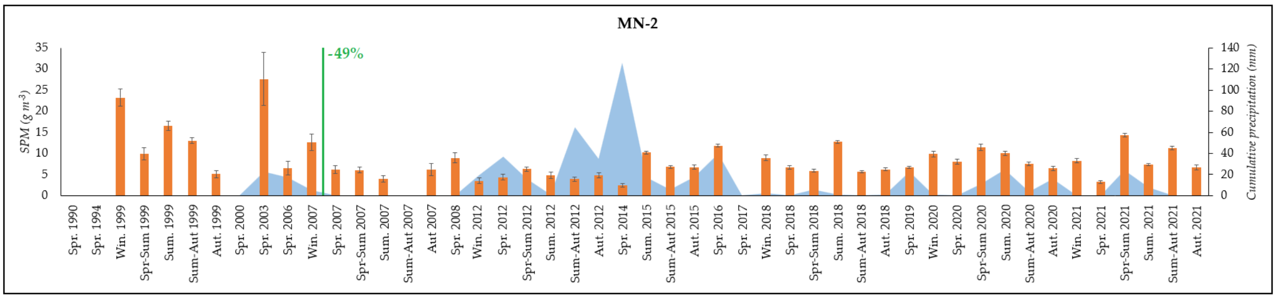

Figure S5. Temporal evolution of SPM concentration (orange columns) of MN-2 pit lake in the MN area (Mantua) in relation to cumulative precipitation (blue polygons) in the 7 days prior to satellite acquisitions. The green line indicates the end of quarrying activities and the mean percentage decrease in SPM concentrations after that event.

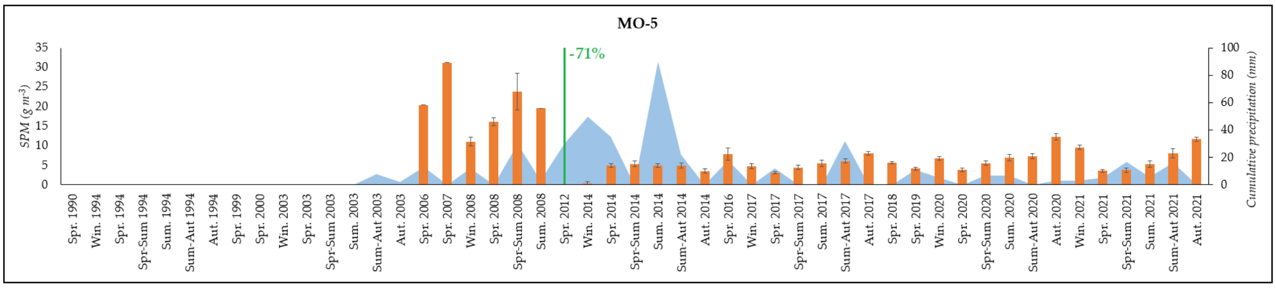

Figure S6. Temporal evolution of SPM concentration (orange columns) of MO-5 pit lake in the MO area (Modena) in relation to cumulative precipitation (blue polygons) in the 7 days prior to satellite acquisitions. The green line indicates the end of quarrying activities and the mean percentage decrease in SPM concentrations after that event.

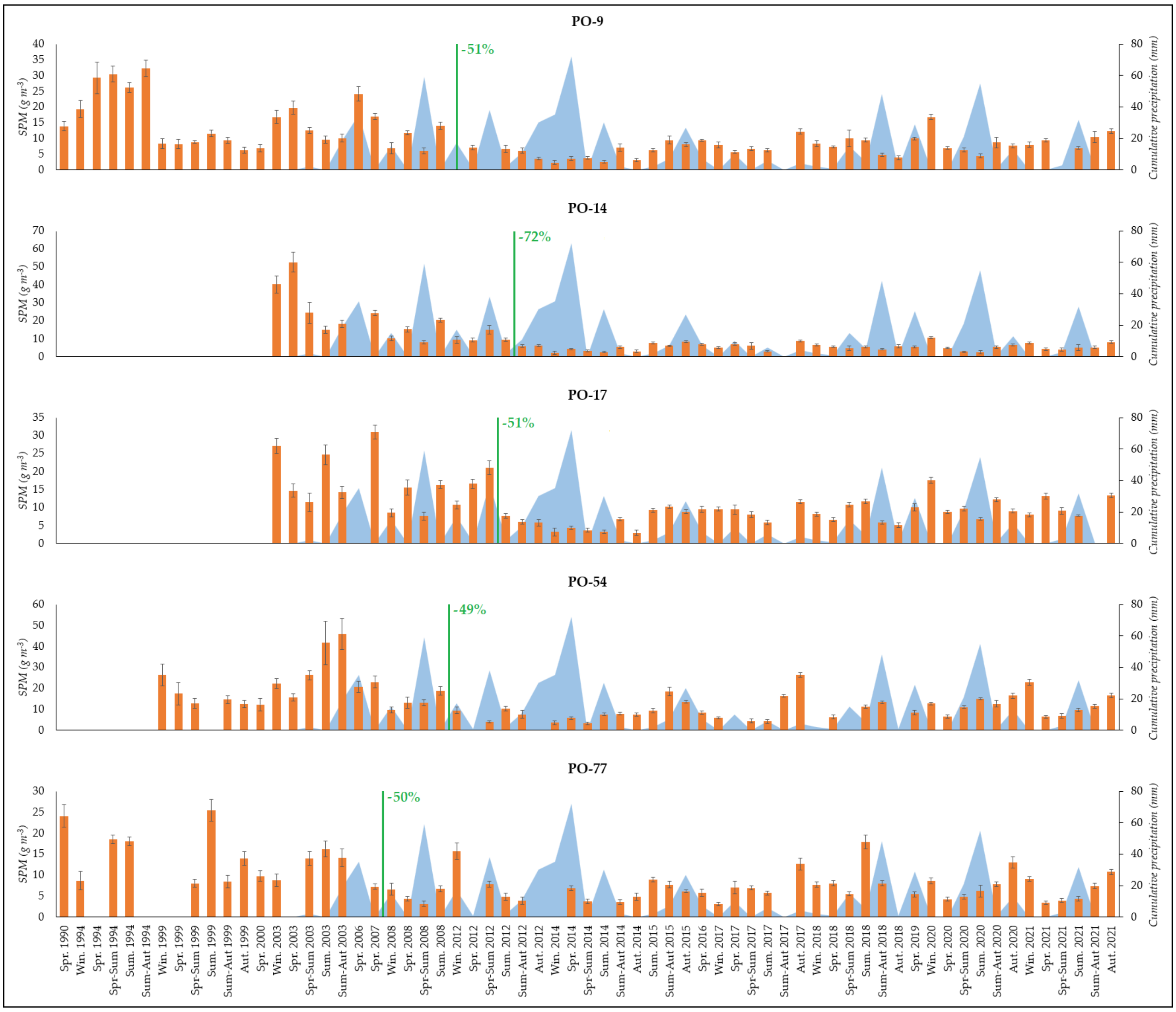

Figure S7. Temporal evolution of SPM concentration (orange columns) of some pit lakes in the PO area (Po River shaft) in relation to cumulative precipitation (blue polygons) in the 7 days prior to satellite acquisitions. The green line indicates the end of quarrying activities and the mean percentage decrease in SPM concentrations after that event.

Figure S8. Temporal evolution of SPM concentration in a pit lake (PO-24, coordinates: 45.057716 N, 10.043437 E) directly connected to river affected by a flood event that occurred between 2018-10-24 and 2018-11-08.

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Chapter IV: Detection of inorganic and organic fraction of SPM concentrations through PRISMA hyperspectral imagery

4.1 Introduction and aim of the study

Suspended Particulate Matter (SPM) is a key parameter for describing water characteristics and plays a crucial role in investigating the quality status of aquatic ecosystems (Pozdnyakov et al., 2005; Xing et al., 2013; Doxaran et al., 2014; Chen et al., 2016; Liu et al., 2017; Zhao et al., 2018; Ciancia et al., 2020; Jiang et al., 2021). This parameter affects several aspects of aquatic environments both in direct and indirect ways, such as water transparency for the first, and primary productivity, nutrient dynamics, aquatic vegetation distribution and light-dependent aquatic organisms for the second one. (Duan et al., 2009; Cao et al., 2017; Gernez et al., 2014; Moore et al., 1997; Doxaran et al., 2002; Havens, 2003; Guildford et al., 2007; Zhu et al., 2013; Hou et al., 2017; Jiang et al., 2021; Ma et al., 2021; Wen et al., 2022). SPM is commonly divided into inorganic (SPIM) and organic (SPOM) fractions. These two components originate from different sources and act differently on optical properties and/or water quality. SPIM refers to the inorganic fraction, mainly composed of minerals, sediment, sand, and other inorganic materials, while SPOM refers to the organic fraction, mainly composed of algae, bacteria, vegetation fragments, dissolved substances, and other biological particles. SPIM is often associated with the larger particles that can sediment more easily, while SPOM is frequently associated with the smaller, less heavy particles that are therefore more easily subject to resuspension and transport. The distinction between these two fractions is critical for understanding changes in the transparency and water quality of an aquatic ecosystem (Liu et al., 2019).
Similar to the estimation of SPM concentration, its partitioning into SPIM and SPOM requires laboratory measurements. However, this approach, in addition to *in situ* measurement campaigns, is unable to comprehensively represent the actual state of the entire water body considered. Therefore, there is a growing need to adopt also remote sensing-based methods for detecting these two parameters in conjunction with laboratory analyses. To monitor SPIM and SPOM separately, it is essential to understand the Inherent Optical Properties (IOP) of the water body. In particular, it is necessary to consider the absorption ($a(\lambda)$) and backscattering ($b_b(\lambda)$) coefficients of water and its constituents. These parameters play a crucial role in the estimation of Optically Active Constituents (OACs), which includes suspended sediments. Absorption properties of suspended particles are fundamental for determining the remote sensing signal and discriminating between organic and inorganic substances (Bricaud et al., 2010; Xue et al., 2017). In addition, these absorption properties are frequently related to the water constituent's concentration, such as phytoplankton and non-algal particles (NAPs) (Shi et al., 2013; Xue et al., 2017; Ylöstalo et al., 2014). Therefore, gaining detailed knowledge of absorption variations in relation to environmental factors is an essential prerequisite for parameterizing bio-optical models, through which SPIM and SPOM can be estimated.
Recently, two studies addressing the detection of inorganic and organic components of SPM using remote sensing techniques have been published. In the study conducted by Liu et al., (2019), both absorption properties and different sources of SPM were considered in order to develop a workflow for simultaneously estimating SPIM and SPOM using multispectral satellite imagery: Sentinel-3 (21 bands in the VIS-NIR domain and a spatial resolution of 300 m) and MODIS (19 bands in the VIS-NIR domain and a spatial resolution ranging from 250 to 1000 m depending on the band). Xiong et al., (2019) exploited the existing correlation with SPIM to estimate through MODIS multispectral imagery the concentration of phosphorus (P), a key parameter for assessing

biological growth and the occurrence of eutrophication in lakes. In both cases, the use of multispectral satellite data was a promising methodology to monitor and estimate the concentration of SPM and its organic and inorganic fractions.
Unlike the studies just mentioned that employ sensors with a coarser spatial resolution but a greater amount of bands in the VIS-NIR domain, Sentinel-2 (S2), which is widely employed in this thesis project, has a higher spatial resolution but, disposes of a limited number of bands that do not allow effective partitioning of SPM concentration into its two main components. Therefore, the main aim of this chapter is to estimate the percentage of SPIM and SPOM in a subsample of pit lakes (PLs), exploiting the hyperspectral satellite PRISMA (PRecursore IperSpettrale della Missione Applicativa) imagery, operated by ASI (Agenzia Spaziale Italiana). PRISMA is a satellite placed in a heliosynchronous orbit, with a revisit time of 29 days that can be reduced to 7 days due to its off-nadir pointing capability. The satellite features a hyperspectral sensor operating in the range of 400 to 2500 nm (239 bands) with a spatial resolution of 30 m (Lopinto and Ananasso, 2013; Coppo et al., 2020; Cogliati et al., 2021). There are several PRISMA products ranging from Level 0 (L0) to Level 2 (L2). For this study, L1 (calibrated and radiometrically corrected radiance) and L2c (geolocated reflectance, obtained through the MODTRAN v.6.0 atmospheric correction processor) were used.

4.2 Materials and methods

4.2.1 *Dataset*

Starting from the eight subsample areas used in Ghirardi et al., (2023b), a sample of PLs was selected as physico-chemical and morphological characteristics are known together with the surrounding land use. Since no *in situ* data synchronous to the satellite passage concerning the percentage of SPIM and SPOM were available, the dominant component of the SPM of each lake was hypothesized based on quarrying activity, the presence of riparian vegetation, and the number of agricultural fields in a buffer zone in their surroundings (Ghirardi et al., 2023a). The selected lakes are located within the following areas: Po and Orba River Park (OR), Po river (PO) and Mantua (MN). These areas were chosen based on the availability of an archive of PRISMA images, as they are the only ones covered by the sensor orbit. Once the areas of interest were identified, the satellite images were downloaded. Specifically, all cloud-free PRISMA images that had corresponding and synchronous S2 images (maximum discrepancy of 2 days) were chosen. Table 3 shows all the information about the dataset used, both for satellite images and PLs.

Table 3. Dataset related to satellite images and pit lakes (PLs) analysed. The code for each lake refers to the geographical area: OR (Po and Orba River Park), PO (Po river) and MN (Mantua). Quarrying activity: C (Ceased) and A (Active). SPM concentration based on the results obtained from Sentinel-2 images: low (0-5 gm^{-3}), medium (5-15 gm^{-3}), high (15-25 gm^{-3}), and very high (> 25 gm^{-3}). SPM composition based on quarrying activity, riparian vegetation and land use: I (inorganic), O (organic).

PRISMA	Sentinel-2	PLs	Coordinates	Quarrying activity	SPM concentration	SPM composition
2020/05/24	2020/05/26	OR-4	45.15901 N, 8.54987 E	C	Low	I
		OR-10	45.07589 N, 8.56615 E	C	Low	O
2020/11/25	2020/11/24	PO-75	44.88843 N, 11.45179 E	A	Medium	I
		PO-77	44.86168 N, 11.52660 E	C	Low	O
		PO-78	44.87368 N, 11.53358 E	A	Medium	I

2021/06/28	2021/06/30	OR-12	45.07852 N, 8.74516 E	C	Low	I
		OR-13	45.04265 N, 8.76741 E	A	Medium	I
		OR-14	45.09804 N, 8.78358 E	A	Low	I
		OR-15	45.06191 N, 8.81496 E	A	High	I
		OR-16	45.07003 N, 8.82942 E	A	Low	I
		OR-17	45.01871 N, 8.82424 E	C	Low	O
		OR-18	45.01737 N, 8.82653 E	C	Low	O
		OR-19	45.03520 N, 8.83447 E	C	High	O
		OR-20	45.03575 N, 8.84018 E	C	Medium	O
		OR-22	45.03093 N, 8.87447 E	C	Low	O
		OR-25	45.07011 N, 8.89687 E	C	Low	O
		OR-26	45.05825 N, 8.92357 E	A	High	I
2021/07/09	2021/07/10	OR-12	45.07852 N, 8.74516 E	C	Low	I
		OR-13	45.04265 N, 8.76741 E	A	Medium	I
		OR-15	45.06191 N, 8.81496 E	A	Medium	I
		OR-16	45.07003 N, 8.82942 E	A	Low	I
		OR-17	45.01871 N, 8.82424 E	C	Low	O
		OR-18	45.01737 N, 8.82653 E	C	Low	O
		OR-19	45.03520 N, 8.83447 E	C	Medium	O
		OR-20	45.03575 N, 8.84018 E	C	Medium	O
		OR-22	45.03093 N, 8.87447 E	C	Low	O
		OR-25	45.07011 N, 8.89687 E	C	Low	O
		OR-26	45.05825 N, 8.92357 E	A	Medium	I
2021/09/23	2021/09/23	OR-1	45.16726 N, 8.48318 E	C	Low	O
		OR-4	45.15901 N, 8.54987 E	C	Low	I
		OR-6	45.14085 N, 8.49449 E	C	Low	O
		OR-8	45.10937 N, 8.52070 E	A	Medium	I
		OR-10	45.07589 N, 8.56615 E	C	Low	O
		OR-11	45.06505 N, 8.58280 E	C	Medium	O
		OR-12	45.07852 N, 8.74516 E	C	Low	I
		OR-13	45.04265 N, 8.76741 E	A	Medium	I
		OR-14	45.09804 N, 8.78358 E	A	Low	I
2021/10/27	2021/10/28	MN-1	45.27207 N, 10.66549 E	C	High	O
		MN-4	45.24021 N, 10.70126 E	C	Medium	O
		MN-5	45.23404 N, 10.66209 E	C	High	O

		MN-8	45.22360 N, 10.67440 E	C	Medium	O
		MN-10	45.21671 N, 10.66502 E	C	Medium	O
		MN-16	45.20608 N, 10.69030 E	C	Medium	O
		MN-20	45.18486 N, 10.73665 E	C	Medium	I
		MN-22	45.17562 N, 10.74095 E	C	Low	O
2021/12/13	2021/12/12	MN-1	45.27207 N, 10.66549 E	C	Low	O
		MN-3	45.24499 N, 10.70713 E	C	Low	O
		MN-4	45.24021 N, 10.70126 E	C	Low	O
		MN-5	45.23404 N, 10.66209 E	C	Low	O
		MN-10	45.21671 N, 10.66502 E	C	Low	O
		MN-13	45.21550 N, 10.67224 E	A	Very high	I
		MN-16	45.20608 N, 10.69030 E	C	Medium	O
		MN-17	45.20796 N, 10.69151 E	C	Low	O

4.2.2 *Processing of satellite images*

Once the synchronous image pairs were identified, seven PRISMA images (both as L1 and L2c, http://prisma.asi.it/) and seven S2 images (L1 only, https://catalogue.onda-dias.eu/catalogue/) were downloaded. All PRISMA images were converted from the original HDF-EOS5 (Hierarchical Data Format - Earth Observing System) file to band sequential file format using ENVI v.5.7 (L3Harris Technologies, USA). L2c were spatially and spectrally resized. Specifically, 36 bands in the VIS-NIR domain (439-744 nm) were maintained. In addition, we eliminated the first bands in the blue domain as they are characterised by very low SNR (Signal to Noise Ratio) (Braga et al., 2022; Pellegrino et al., 2023), while the bands above 744 nm were not necessary for our purposes.
After that, they were converted to Remote Sensing Reflectances (Rrs) by dividing by π. To obtain the same product from L1, an atmospheric correction was performed using ACOLITE (dark spectrum fitting (DSF) algorithm) (Vanhellemont and Ruddick, 2018). In particular, L1 was used as input, while L2c was used to extract the viewing geometries (i.e., sun and view zenith and relative azimuth angles). For more information on processing of PRISMA images with ACOLITE, see Braga et al., (2022).
S2 L1 images were also atmospherically corrected using ACOLITE and SPM concentrations were extracted using the SPM_Nechad2016 algorithm. This process was conducted following the same methodology described in Ghirardi et al., (2023b). For both PRISMA and S2 images, specific Regions of Interest (ROIs) were created for each PLs listed in Table 3, avoiding pixels too close to the shoreline. Since PRISMA images synchronous to the measurement campaigns carried out in recent years on various PLs were not available, we used the spectral signatures and SPM concentrations obtained from S2 L1 images as ground truth since a strong agreement between these satellite products and those measured *in situ* was demonstrated in Ghirardi et al., (2023b).
To obtain the SPM concentrations and the percentage of SPIM and SPOM from PRISMA images, the BOMBER (Bio-Optical Model-Based tool for Estimating water quality and bottom properties from Remote sensing images) bio-optical model was adopted (Giardino et al., 2012). BOMBER is a non-linear optimization algorithm used to estimate the concentration of water quality parameters (e.g., SPM and Chl-a), and for substrate coverage classification and bathymetry mapping. Rrs

PRISMA L1 products were used as input to estimate the percentage of SPIM and SPOM from SPM concentration. The bio-optical model was applied in the shallow water mode, and a mask built from the previously created ROIs was used for each image. Specifically, based on the hypothesized SPM composition for each lake, the BOMBER code was parameterized in two different modes. Figure 16 shows the spectra for the Specific Inherent Optical Properties (SIOPs) of the water constituents for both "inorganic" and "organic" PLs.

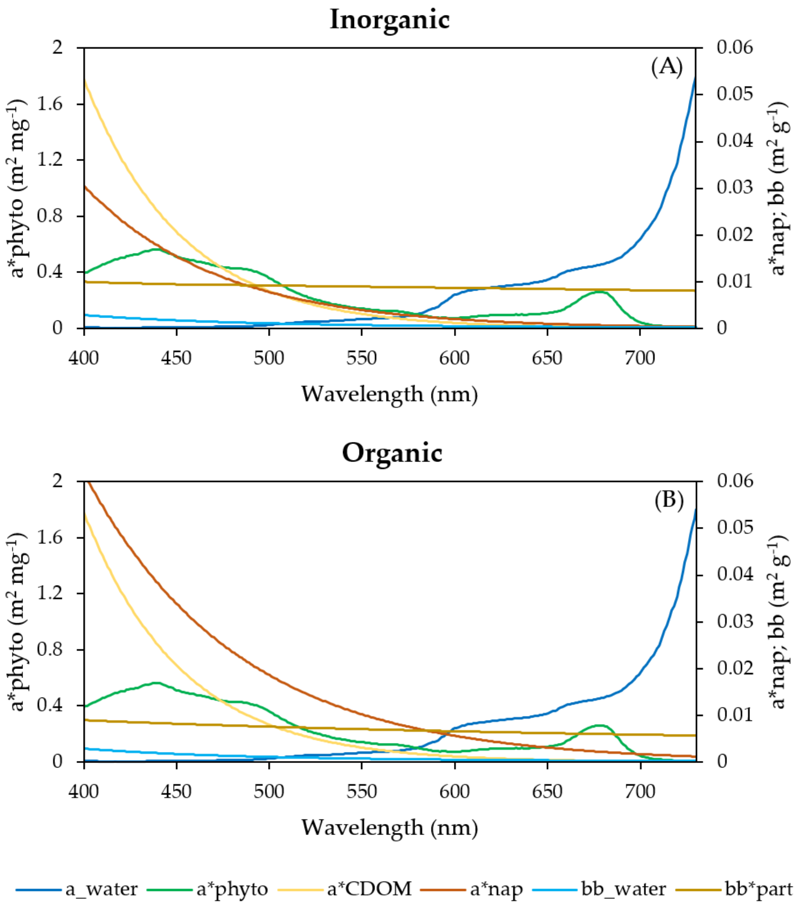

Figure 16. (A) The SIOPs spectra used to parameterize the BOMBER for "inorganic" pit lakes. (B) The SIOPs spectra used to parameterize the BOMBER for "organic" pit lakes. "a_water": absorption of water; "a*phyto": absorption of phytoplankton; "a*CDOM": absorption of colored dissolved organic matter; "a*nap": absorption of non-algal particle; "bb_water": backscattering of water; "bb_part": backscattering of total particulate matter.

The absorption of non-algal particle (aNAP) spectra used to parameterize the BOMBER code were obtained from data collected *in situ* in some PLs located along the Po River during the measurement campaigns described in Ghirardi et al., (2023b). The PLs considered were characterised by very similar SPM concentrations ranging from 4 to 10 gm^{-3} (Fig. 17). Water samples were filtered with a glass fiber filter (Whatman, GF/F) at low vacuum. The absorption spectrum of particles retained on the filter was determined using the filter pad technique (see Babin et al., 2003). Absorbance was measured between 380 and 750 nm with 1-nm increments using a dual beam spectrophotometer (Perkin Elmer, Lambda 35). The absorption coefficient of NAP was determined after pigment bleaching with acetone. The following expression was fitted to the aNAP(λ) spectra:

Eq. 11 $a_{NAP}(\lambda) = \hat{a}_{NAP}(\lambda_r) e^{(-S_{NAP}(\lambda-\lambda_r))}$

where $\hat{a}_{NAP}(\lambda_r)$ is the absorption estimate at reference wavelength. The fit was done for data between 380 and 730 nm, excluding the 400-480 and 620-710 nm ranges to avoid any residual pigment absorption that might still have been present after acetone treatment.
Figure 17 clearly highlights how aNAP spectra can be divided into two macro categories representing PLs dominated by "inorganic" or "organic" fraction of SPM. Specifically, the spectra for the "organic" PLs show a higher absorption peak in the blue domain compared to the "inorganic" ones.

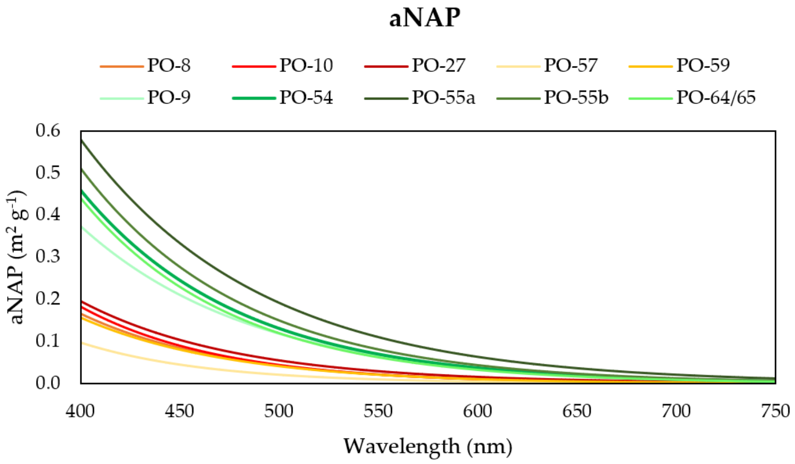

Figure 17. Absorption spectra measured in lab from *in situ* samples of the particle fraction of water some pit lakes (PLs) located near the Po River. The "organic" PLs are represented with green shades, while the "inorganic" PLs are represented with yellow-red shades.

4.2.3 *Statistical analysis*

After the processing of all satellite images, a comparison of the mean spectral signatures of each PLs listed in Table 3 was conducted. Specifically, the Rrs obtained from S2 images was compared with those obtained from PRISMA images (both L1 and L2c). The main aim of this comparison was to determine which PRISMA level gave the most similar spectral signature to S2 products, to identify the most appropriate level to be used as input for the BOMBER model.
The first six bands of S2 (B1=443 nm, B2=490 nm, B3=560 nm, B4=665 nm, B5=705 nm, B6=740 nm) were compared with the six PRISMA bands with wavelengths most similar to those of S2 (B6=446 nm, B12=490 nm, B21=559 nm, B32=665 nm, B37=704 nm, B41=744 nm). A series of statistical indices, including coefficient of determination (R^2), Mean Absolute Error (MAE), Root Mean Square Error (RMSE), Mean Absolute Percentage Error (MAPE), and Spectral Angle (SA) were calculated. For the formulas, see paragraph 3.2.3 of this thesis (Eq. 6-10).
The BOMBER code was run with the aim of retrieving SPM concentrations and percentages of SPIM and SPOM with the better PRISMA level identified. The resulting SPM products were then compared with those derived from the S2 images employing the same metrics used previously (with the exception of SA). Finally, the obtained percentages of SPIM and SPOM were compared with the hypothesized SPM composition of each lake.

4.3 Results and discussion

The comparisons between the spectral signatures of S2 and those of PRISMA (both L1 and L2c) highlighted a good correlation in the visible spectrum, while some issues emerged in the NIR domain. In particular, as previously observed in Ghirardi et al., (2023b), the NIR band (740 nm) performs much better in turbid than in clear waters. This phenomenon is probably attributable to an error in atmospheric correction due to the intrinsic properties of water in the infrared domain. Considering the metrics used (R^2, MAE, RMSE, and MAPE), it emerged that the first two bands of S2 (443 and 490 nm) are better represented by PRISMA L2c, while the next four bands (560, 665, 705, and 740 nm) are much more similar to those of PRISMA L1. On the other hand, considering the entire spectral signature, the SA value corresponding to the comparison between S2 and PRISMA L2c is higher than the comparison between S2 and PRISMA L1 (14.5° ± 6.9° and 11.7° ± 6.3°, respectively). Based on these results, the Rrs products obtained from PRISMA L1 were used as input for the BOMBER model, with the aim of estimating the concentrations of SPM and the respective fractions of SPIM and SPOM. The graphs for the statistical metrics are shown in Figure 18, while some example of spectral signatures are shown in Figure S10.

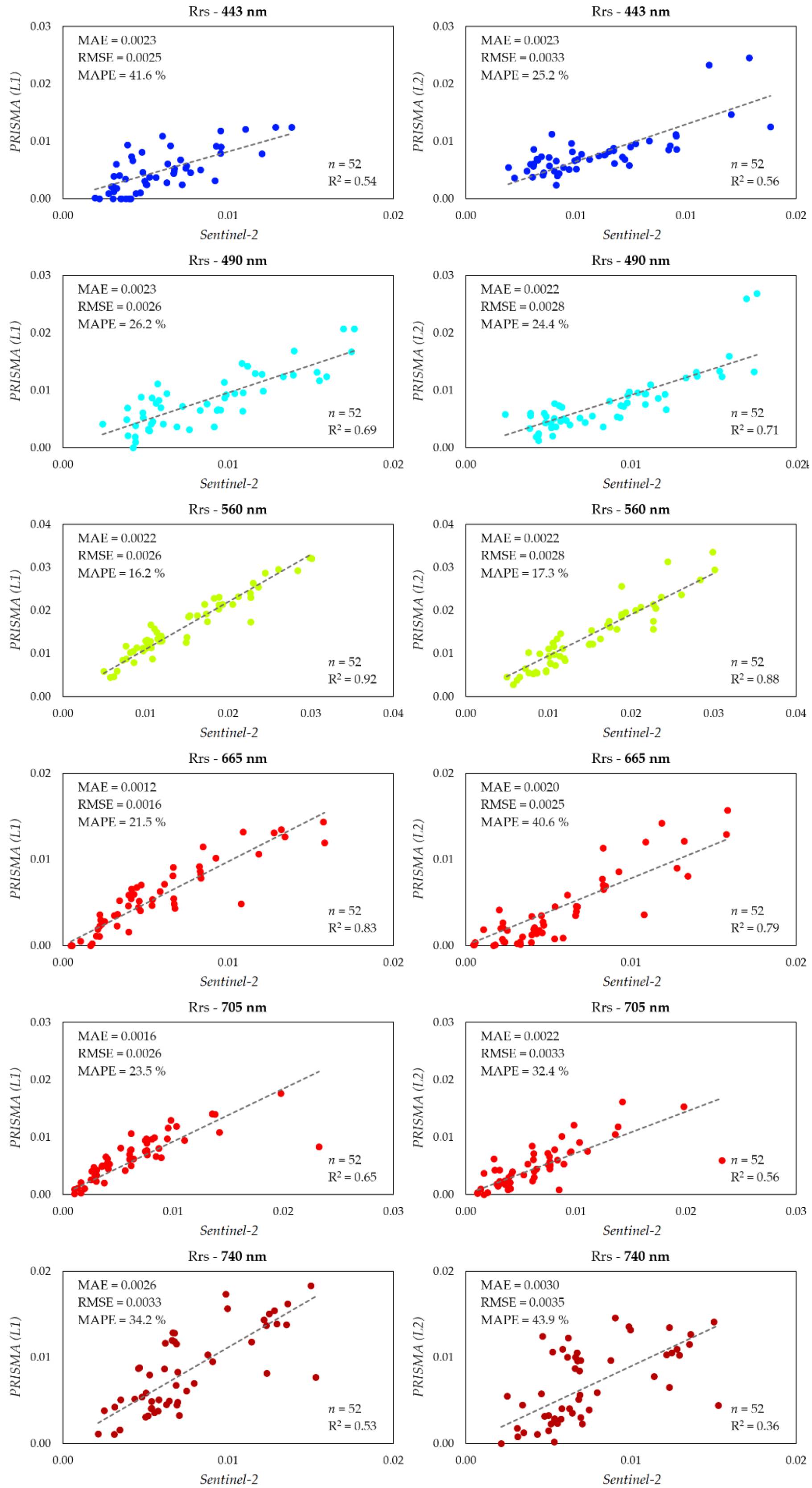

Figure 18. Scatterplots between Sentinel-2 Rrs and PRISMA Rrs (L1 and L2c). On the left the comparison between S2 and PRISMA L1, while on the right the comparison between S2 and PRISMA L2c. The colored dots represent the first six bands of S2 and the six most similar bands of PRISMA. “n” represents the sample size, “R^2” the coefficient of determination, “MAE” the mean absolute error, “RMSE” the root mean square error, “MAPE” the mean absolute percentage error, and the dashed gray lines refers to the regression lines.

Comparison of SPM concentrations obtained by ACOLITE (S2 images) and those obtained by BOMBER (PRISMA L1 images) showed very good agreement (Fig. 19). From the scatterplot, the data related to a very turbid lake (MN-13) was removed, because although there was good correlation (43.4 ± 6.4 gm^{-3} and 39.4 ± 2.7 gm^{-3}), it would have over-improved the R^2 value.

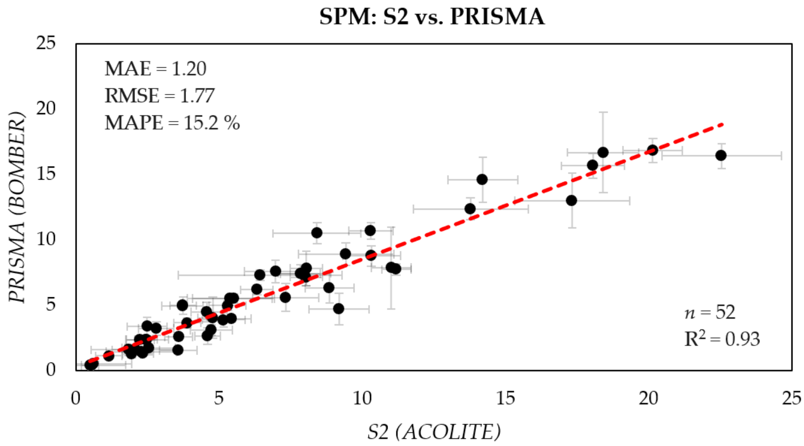

Figure 19. Scatterplots between the SPM concentration derived from S2 (ACOLITE) and those obtained from PRISMA L1 (BOMBER). "n" represents the sample size, "R^2" the coefficient of determination, "MAE" the mean absolute error, "RMSE" the root mean square error, "MAPE" the mean absolute percentage error. The gray lines represents standard deviations, while the dashed red line refers to the regression lines.

Table 4 shows the SPM concentrations obtained through the two methods previously described, together with the percentages of SPIM and SPOM obtained via BOMBER, using PRISMA L1 as input. Comparison of SPIM and SPOM percentages with the hypothesized SPM composition showed that the bio-optical model was able to correctly estimate the two SPM fractions in the majority of PLs (specifically, 47 out of 53). Where disagreement was found, the fractions obtained were in equilibrium with values close to 50%. The results also show that, as expected, active PLs exhibit a higher percentage of SPIM, although it rarely exceeds 60%. This phenomenon could be attributed to the presence of riparian vegetation, but mainly to the influence of agricultural fields surrounding all the PLs analysed. This condition leads to multi-source inputs that result in a lack of sharp distinctions between the two fractions of SPM, generating in many cases an equilibrium with a slight tendency toward one of the two fractions. The only area the dominance of SPOM is observed is MN. In this area, with the exception of lake MN-13, quarrying activity has ceased in all PLs and they are surrounded by agricultural fields. As a result, the organic fraction was higher than the inorganic one.

Table 4. SPM concentrations (mean ± st.dev.) obtained from PRISMA L1 and Sentinel-2 through the BOMBER bio-optical model and ACOLITE neural network, respectively. For the PRISMA data, also the percentages of SPIM (Suspended Particulate Inorganic Matter) and SPOM (Suspended Particulate Organic Matter) derived from the SPM concentration. In red the percentages of SPIM and SPOM that mismatched expectations based on SPM composition. The code for each lake refers to the geographical area: OR (Po and Orba River Park), PO (Po river) and MN (Mantua). SPM composition based on quarrying activity, riparian vegetation and land use: I (inorganic), O (organic).

PRISMA	**Sentinel-2**	**PLs**	**SPM** *[g m^{-3}]* (PRISMA)	**SPM** *[g m^{-3}]* (Sentinel-2)	**SPM composition**	**SPIM** *[%]*	**SPOM** *[%]*
2020/05/24	2020/05/26	OR-4	1.1 ± 0.1	1.2 ± 0.5	I	67	33
		OR-10	2.4 ± 0.3	2.5 ± 0.7	O	46	54

2020/11/25	2020/11/24	PO-75	8.8 ± 0.8	10.3 ± 1.0	I	56	44
		PO-77	1.6 ± 0.4	1.8 ± 0.6	O	43	57
		PO-78	12.4 ± 0.8	13.8 ± 2.0	I	55	45
2021/06/28	2021/06/30	OR-12	2.3 ± 0.3	2.2 ± 0.5	I	54	46
		OR-13	8.9 ± 0.9	9.4 ± 1.7	I	53	47
		OR-14	1.6 ± 0.4	2.1 ± 1.6	I	49	51
		OR-15	16.4 ± 1.0	22.5 ± 2.1	I	52	48
		OR-16	1.5 ± 0.3	2.3 ± 1.2	I	53	47
		OR-17	2.7 ± 0.6	4.6 ± 0.4	O	55	45
		OR-18	2.6 ± 0.3	3.6 ± 0.5	O	49	51
		OR-19	16.7 ± 3.1	18.4 ±1.2	O	40	60
		OR-20	7.4 ± 0.7	7.8 ± 0.7	O	48	52
		OR-22	1.3 ± 0.2	2.3 ± 0.4	O	50	50
		OR-25	1.8 ± 0.5	2.5 ± 0.9	O	55	45
		OR-26	13.0 ± 2.1	17.3 ± 2.0	I	57	43
2021/07/09	2021/07/10	OR-12	2.4 ± 0.2	2.5 ± 0.4	I	58	42
		OR-13	5.6 ± 1.1	7.3 ± 1.2	I	55	45
		OR-15	7.7 ± 0.5	11.2 ± 0.5	I	53	47
		OR-16	0.4 ± 0.1	0.6 ± 1.2	I	59	41
		OR-17	1.5 ± 0.2	3.6 ± 0.7	O	41	59
		OR-18	1.3 ± 0.3	2.0 ± 0.3	O	40	60
		OR-19	14.6 ± 1.7	14.2 ± 1.2	O	47	53
		OR-20	7.3 ± 0.4	6.4 ± 2.9	O	47	53
		OR-22	0.4 ± 0.1	0.5 ± 0.3	O	40	60
		OR-25	0.5 ± 0.2	0.6 ± 1.3	O	41	59
		OR-26	4.7 ± 1.2	9.2 ± 1.0	I	56	44
2021/09/23	2021/09/23	OR-1	3.7 ± 0.4	3.9 ± 0.5	O	47	53
		OR-4	5.0 ± 0.7	3.7 ± 0.5	I	59	41
		OR-6	5.5 ± 0.4	5.5 ± 1.4	O	47	53
		OR-8	7.2 ± 0.6	8.0 ± 0.6	I	49	51
		OR-10	5.0 ± 0.3	5.3 ± 0.6	O	47	53
		OR-11	7.4 ± 0.5	8.0 ± 0.9	O	52	48
		OR-12	4.9 ± 0.2	3.7 ± 0.7	I	60	40
		OR-13	6.2 ± 0.4	6.3 ± 0.6	I	49	51
		OR-14	5.5 ± 0.3	5.4 ± 1.5	I	60	40

2021/10/27	2021/10/28	MN-1	16.8 ± 0.9	20.1 ± 1.0	O	37	63
		MN-4	10.5 ± 0.8	8.4 ± 1.5	O	36	64
		MN-5	15.7 ± 1.0	18.0 ± 1.1	O	39	61
		MN-8	7.8 ± 1.3	8.1 ± 0.6	O	38	62
		MN-10	6.3 ± 1.2	8.9 ± 0.9	O	39	61
		MN-16	7.9 ± 5.1	11.0 ± 0.7	O	39	61
		MN-20	10.7 ± 0.7	10.3 ± 0.8	I	51	49
		MN-22	4.0 ± 0.2	5.4 ± 0.7	O	37	63
2021/12/13	2021/12/12	MN-1	4.5 ± 0.8	4.6 ± 0.5	O	39	61
		MN-3	3.4 ± 0.7	2.5 ± 0.7	O	39	61
		MN-4	3.2 ± 0.5	2.8 ± 0.6	O	40	60
		MN-5	3.9 ± 0.6	5.2 ± 0.7	O	40	60
		MN-10	3.1 ± 0.4	4.7 ± 0.7	O	41	59
		MN-13	39.4 ± 2.7	45.4 ± 6.4	I	53	47
		MN-16	7.6 ± 0.9	7.0 ± 1.1	O	49	51
		MN-17	4.0 ± 1.6	4.8 ± 0.5	O	39	61

In conclusion, PRISMA L1 has proven to be a suitable tool for deriving SPIM and SPOM percentages from SPM concentration. Precise knowledge of the composition of suspended solids is essential for a more in-depth assessment of water quality status. In particular, specific algorithms can be developed from the inorganic fraction to estimate phosphorus (P) concentration, while an estimation of chlorophyll-a (Chl-a) concentration can be obtained from the organic fraction, two fundamental parameters for assessing the trophic status of waters.

The next step should involve a more robust validation of the results, through the use of data collected *in situ* synchronous to the satellite acquisition. In addition, it could be advantageous to use the PRISMA panchromatic band to improve the spatial resolution from 30 m to 5 m, thus simplifying the analysis of smaller PLs. However, considering the challenges associated with *in situ* data collection in PLs (many of them are private and not accessible) and the difficulties in planning PRISMA acquisition (there are many requests and they are not always fulfilled), it would be helpful to install fixed instruments that can continuously monitor the spectral signatures of these lakes and derive from them all the water quality parameters needed for this type of analysis.

Figure S9. True color images and SPM concentration maps for some pit lakes (PLs). The mean ± st.dev. of SPM concentration estimated from Sentinel-2 (via ACOLITE) and PRISMA (via BOMBER) images, are shown for each PLs. For PRISMA images, the composition of the inorganic (SPIM, in red) and organic (SPOM, in green) fractions of the suspended solids is also represented.

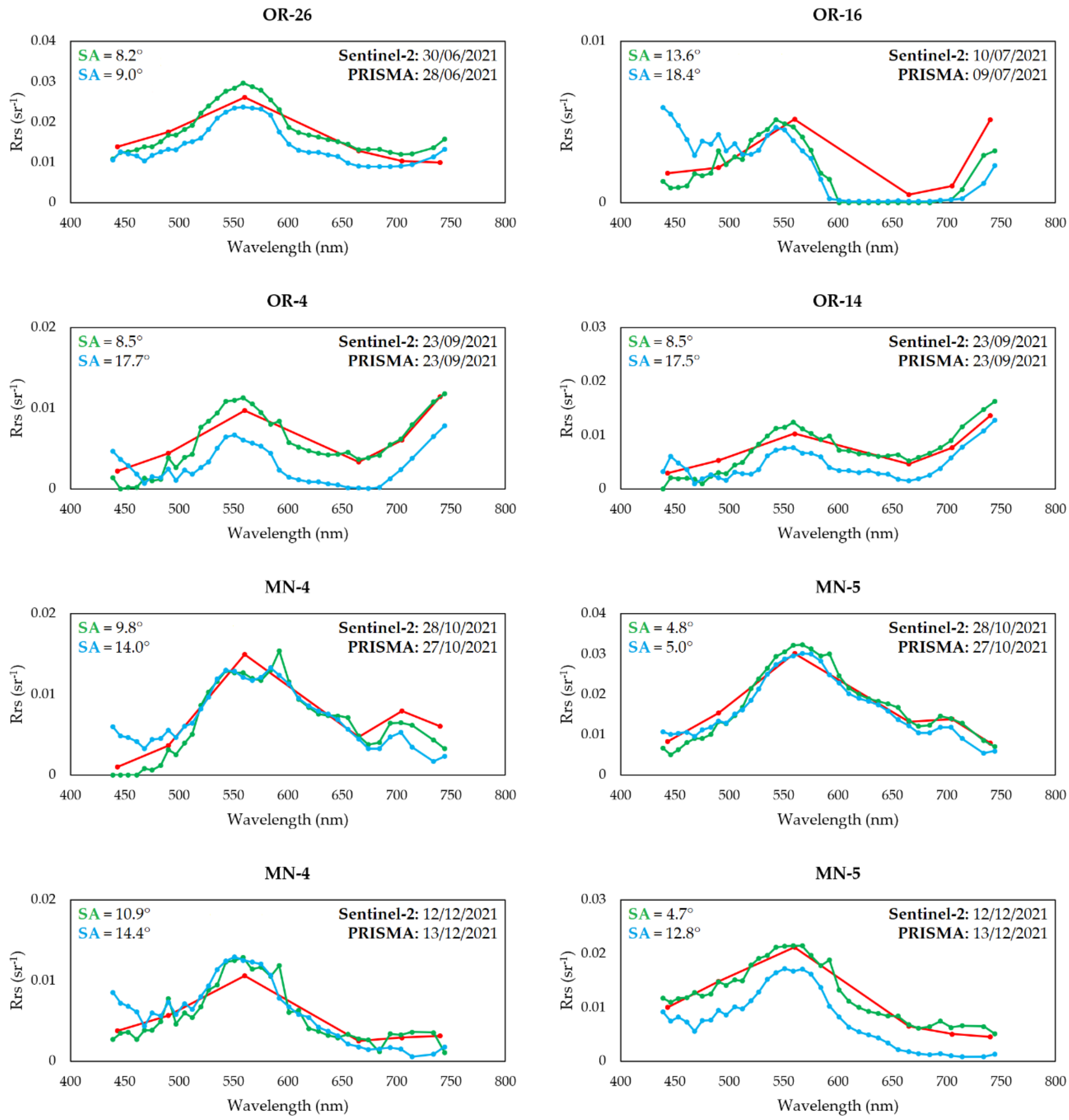

Figure S10. Examples of spectral signatures for some pit lakes analysed. In red are the mean spectral signatures obtained from Sentinel-2 (S2) images, in green those from PRISMA L1 images, and in blue those from PRISMA L2c images. Displayed for each lake are the acquisition dates and SA (Spectral Angle) values for comparisons between S2 and PRISMA spectral signatures.

4.5 References

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Chapter V: Conclusions and future development

This Ph.D. project confirmed that the application of remote sensing offers a new perspective for the identification, quantification, classification, and spatio-temporal analysis of pit lakes (PLs) at the basin-scale. The reliability of Landsat (L5 and L7) and Sentinel-2 (S2) satellites in performing basin-scale assessments of SPM and λ_{dom} concentrations in small and dynamic inland waters such as PLs has been confirmed. At the same time, the PRISMA satellite has emerged as a suitable tool for estimating organic and inorganic fractions of suspended solids. The approach adopted in this study enabled a comprehensive and in-depth evaluation of the current state of PLs, as well as the evolution of these artificial aquatic systems over three decades (1990 to 2021). The spatial and spectral characteristics of these satellites have opened new horizons in the study of these lakes, which were otherwise inaccessible using traditional limnological techniques. SPM concentration and λ_{dom}, being two parameters intrinsically related to the optical characteristics of water, have been used as preliminary indicators to assess the water quality status of PLs. However, despite their importance, they are not exhaustive in defining the quality of these aquatic systems.

The key results of this study include that the water quality of these lakes depends on multiple factors, including their location and size. It was demonstrated that the PLs with a poor water quality (based on both SPM concentration and λ_{dom}) were those of small sizes, often located near or directly connected to rivers. In addition, quarrying activity appears to have a significant impact, as it has been demonstrated that the end of such activities results in a marked decrease in SPM concentrations. Conversely, the extraction of inert materials from the lake bottom causes an increase in SPM concentrations, an effect that propagates rapidly throughout the water body from the point of extraction.

The classification of all the small lentic waters typologies in the Po River basin allowed the comparison between the PLs and W&Os. PLs were more numerous and more widely distributed than W&Os, especially in the most urbanized zones where PLs are almost the only lentic water bodies. In this context, the advantage of their high number and distribution, as well as their significant potential for nitrogen removal, these lentic water bodies can be considered suitable for rehabilitating aspects of the riverscape and addressing both the water scarcity and the nitrate contamination of surface waters. Progressive water storage in these lentic water bodies could provide at least a transient buffer against drought, whose extent will depend upon the stored volume. It has been estimated that the total water volume in the PLs of the Po River basin is comparable to the total exploitable water volume in the main Apennine reservoirs of the basin. In particular, the proximity of PLs to rivers makes them subject to flood pulses. This allows them to act as a retention system, making water available for days after the flood ceases. These flood events affect the availability of oxygen in the water column and consequently also the main biogeochemical processes. Therefore, PLs not only act as water storage containers, but also behave as metabolic reactors. In this regard, they are characterized by among the highest denitrification rates in the literature. However, nitrogen removal rates depend on many factors, such as the extent of shallow waters versus the hypolimnetic sediments. To address such limitations, it would be desirable to integrate ecological criteria in quarry design in advance. For these functions, they deserve attention as possible natural-based solutions and, specifically, as Natural Water Retention Measures (NWRM) for managing and rehabilitating altered riverscapes (http://nwrm.eu/).

An important future step in their study might be the more detailed calculation of the actual volume of PLs and especially the assessment of the extent of shallow waters, which are the most reactive part of the lakes. In addition, considering the challenges associated with *in situ* data collection in PLs (many of them are privately owned and obtaining permits is complex) and the limitation associated with the optical remote sensing, the installation of fixed instruments for collecting in continuous spectral measurements for estimating water quality parameters could represent a turning point. This type of approach would allow for further insight into the study of water quality through continuous measurements that could overcome the limitations of satellite remote sensing, such as revisit time and cloud cover. The integration of *in situ* and remote data both radiometric and limnological can be fundamental in the knowledge of the bio-geochemical evolution of these artificial aquatic systems.

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