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Earth observation for monitoring and mapping of cyanobacteria blooms. Case studies on five Italian lakes

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ABSTRACT

Cyanobacterial blooms occur in many parts of the world as a result of entirely natural causes or human activity. Due to their negative effects on water resources, efforts are made to monitor cyanobacteria dynamics. This study discusses the contribution of remote sensing methods for mapping cyanobacterial blooms in lakes in northern Italy. Semi-empirical approaches were used to flag scum and cyanobacteria and spectral inversion of bio-optical models was adopted to retrieve chlorophyll-a (Chl-a) concentrations. Landsat-8 OLI data provided us both the spatial distribution of Chl-a concentrations in a small eutrophic lake and the patchy distribution of scum in Lake Como. ENVISAT MERIS time series collected from 2003 to 2011 enabled the identification of dates when cyanobacterial blooms affected water quality in three small meso-eutrophic lakes in the same region. On average, algal blooms occurred in the three lakes for about 5 days a year, typically in late summer and early autumn. A suite of hyperspectral sensors on air- and space-borne platforms was used to map Chl-a concentrations in the productive waters of the Mantua lakes, finding values in the range of 20 to 100 mgm^{-3}. The present findings were obtained by applying state of the art of methods applied to remote sensing data. Further research will focus on improving the accuracy of cyanobacteria mapping and adapting the algorithms to the new-generation of satellite sensors.

Key word: Remote Sensing; monitoring; chlorophyll-a; lakes; hyperspectral.

INTRODUCTION

In the past two centuries, human activity has altered the global climate to such an extent that this historical period has been called the Anthropocene (Steffen *et al.*, 2007). The effects of climate change are threatening not only water safety and accessibility but also the quality of aquatic ecosystems, leading to loss of biodiversity (Bálint *et al.*, 2011; Harley, 2011), altered of reproductive cycles, and invasion of allochthonous species (Dukes and Mooney, 1999; Walther *et al.*, 2009). Deteriorating aquatic ecosystems may have a strong economic impact (Landsberg, 2002; Falconer and Humpage, 2005; Backer and McGillicuddy, 2006): Dodds *et al.* (2013) calculated that freshwater ecosystems disruption due to human activity reduced the value of marketable aquatic benefits by ~16% globally, or ~$900 billion. Future climate change scenarios predict rising air and water temperatures, enhanced vertical stratification of aquatic ecosystems, and changing seasonal and annual weather patterns. Climate models foresee more frequent and more intense rainfall events (with storms and floods) alternating with longer periods of drought (Dokulil *et al.*, 2009; Dokulil and Teubner, 2011). Such conditions are ideal for the growth, dominance, persistence, and geographic expansion of several harmful cyanobacteria species (Paerl and Huisman, 2009; Reichwaldt and Ghadouani, 2012). Global climate change and anthropic eutrophication are expected to accelerate the shift to turbid water and cyanobacteria-dominated conditions in aquatic environments (Jöhnk *et al.*, 2008; Paerl and Huisman, 2008; Slim *et al.*, 2014; Rousseaux and Gregg, 2015). The optimal water temperature for the growth of cyanobacteria (>25°C; Robarts and Zohary, 1987; Coles and Jones, 2000) is higher than that of green algae or diatoms (Wetzel, 2001). The density of water drops proportionally to its increase in temperature contributing to its vertical stratification in aquatic ecosystems and promoting cyanobacteria growth in the epilimnion (Salmaso, 2005; Winder and Sommer, 2012). Global warming may prolong the annual period of water stratification (Markensten *et al.*, 2010), increasing the dominance of cyanobacteria and nitrogen fixation (Elliott, 2012; Hense *et al.*, 2013). Cyanobacteria generally thrive on higher nutrient loads (N, P), while nutrient stoichiometry determines interspecific competition between cyanobacteria and other algae (Savadova, 2014). Changes in physical parameters such as surface water temperature over time can even lead to algal species succeeding one another during the same bloom, as reported by Wu *et al.* (2016; Dianchi Lake, China). Recent studies indicate that cyanobacteria have increased far more than other phytoplankton communities since c. 1800, and especially after 1945 (Taranu *et al.*, 2015). Coupled with the growing demand for water safety, this trend will probably pose critical environmental and socio-economic problems in the next few years (Paerl and Paul, 2012). Australia, Canada, some European countries and the United States have started state-run toxin monitoring programs (including some for freshwaters) and applied marine fish and shellfish harvesting restrictions, but studies on freshwater harmful algal blooms (HABs), including cyanobacteria and cyanotoxins, lags far behind research on marine HABs and their biotoxins (Carmichael, 2001). Most countries have small research programs on freshwater HABs with small budgets, despite cyanotoxins being considered a priority by the European Water Framework Directive and World Health Organization (Chorus, 2005). The globally increasing frequency of HABs has prompted investigations into environmental monitoring methods and protocols (Lopez *et al.*, 2008). Analyses on blooming toxins and their concentrations produce relevant information, but too late for the prevention of health risks, so early-warning tools for continuously monitoring aquatic ecosystems are a strong research priority (Lopez *et al.*, 2008; Fadel *et al.*, 2014).

The typical dynamics of cyanobacterial blooms make monitoring their quantity and spatial/temporal distribution difficult. Surface blooms can appear within hours and without warning, due not to rapid cell growth but to the upward migration of existing dispersed population. Their onset and severity therefore depend partly on the size of the existing which need not to be particularly large, but becomes much more concentrated as it floats to the surface (Oliver and Ganf, 2000). Processing phytoplankton samples is time-consuming, whereas immediate measurements are often needed to ensure the safe use of water resource. Local agencies monitoring water quality have to combine monitoring programs with faster techniques, which may also be used for a synoptic coverage of their water systems. This is where Earth observation (EO) might provide valuable data (Hestir *et al.*, 2015) on sites of algal blooms (Wang and Shi, 2008; Stumpf *et al.*, 2012; Matthews and Odermatt, 2015), or their duration across multiple lakes in a given ecoregion. These data are useful for establishing *in situ* monitoring programs, planning *in situ* sampling activities, and identifying environmental factors that can promote cyanobacterial blooms.

Several studies demonstrated the capability of mapping algal blooms with optical radiometers operated from the ground or on to space-borne platforms (Palmer *et al.*, 2015; Matthews, 2014; Odermatt *et al.*, 2012). Good results were obtained using specific empirical/semi-empirical algorithms for a given site sensor (Matthews *et al.*, 2012; Hu *et al.*, 2010; Kudela *et al.*, 2015; Shi *et al.*, 2015), or physically based approaches based on spectral inversion of analytical/semi-analytical models, or neural networks (Doerffer and Schiller, 2008; Riha and Krawczyk, 2011; Wynne *et al.*, 2010; Li *et al.*, 2013; Mishra *et al.*, 2013; Li *et al.*, 2015), or hybrid solutions (Carvalho *et al.*, 2010; Matsushita *et al.*, 2015). The first and most often adopted approach involves studying the spectral shape of a signal reflected by water in the visible- near-infrared (VIS-NIR) spectral range (Gilerson *et al.*, 2010; Gurlin *et al.*, 2011; Gitelson *et al.*, 2008). Accessory photosynthetic pigments make cyanobacteria distinguishable from other phytoplankton communities based on their typical features in water reflectance spectra (Babin and Stramski, 2002; Pozdnyakov and Grassl, 2003; Roy *et al.*, 2011). Phycocyanin (PC), the diagnostic pigment most often used to detect cyanobacteria (Dekker *et al.*, 1995; Schalles and Yacobi, 2000; Simis *et al.*, 2007; Randolph *et al.*, 2008; Duan *et al.*, 2012; Yacobi *et al.*, 2015), has characteristic absorption and reflectance peaks around 620nm and 650nm, respectively (Gons *et al.*, 2005; Simis *et al.*, 2005). Phycoerythrin (PE), another specific pigment, has absorption and reflectance peaks at 565 nm and 600 nm, respectively (Bresciani *et al.*, 2011).

Multispectral sensors (*e.g.*, Landsat and more recently Sentinel-2) are generally unable to distinguish between waters dominated by cyanobacteria *vis-à-vis* by other algal species because their spectral band configuration is unsuitable for detecting features of PC-related reflectance or other characteristics unique to cyanobacteria. These sensors might be used in spectral inversion techniques (Dekker *et al.*, 1991), however, to map water quality parameters (including Chl-a concentration), and in empirical relations with phytoplankton pigments (Vincent *et al.*, 2004). Ocean color sensors (MERIS from 2002 to 2012 and now Sentinel-3), have bands appropriate for identifying spectral features due to Chl-a and both PC (Becker *et al.*, 2009; Qi *et al.*, 2014; Dash *et al.*, 2011) and PE (Westberry *et al.*, 2005; Bresciani *et al.*, 2011), but not at low concentrations (Kutser *et al.*, 2006), or in small lakes (where a 300-m pixel size is not good enough for image analysis). Ground-based observations like those obtained with hyperspectral sensors can provide reference measures for EO data validation (Brando *et al.*, 2016; Zibordi *et al.*, 2009), mediate between EO, *in situ* and laboratory data (Bresciani *et al.*, 2013) and generate monitoring data for areas too narrow for EO data (Hommersom *et al.*, 2012). Airborne and space hyperspectral sensors provide a contiguous for identifying key water quality indicators and phytoplankton pigments (Hestir *et al.*, 2015). Finally, integration of multi-sensor EO data, such as MERIS and

ASAR imagery (Adamo *et al*., 2013; Bresciani *et al*., 2014) or MERIS and MODIS (Olmanson *et al*., 2011; Shuchman *et al*., 2013; Schaeffer *et al*., 2013) provides further insight on spatial patterns under cloud, or at different times of day. Numerous variables may correlate with aquatic optics data retrievable by remote sensing data, including cyanobacteria cell counts (Hunter *et al.*, 2010), biovolumes (Reinart and Kutser, 2006), pigment concentrations measured by fluorimetry (Giardino *et al*., 2010; Seppala *et al.*, 2007) and high-performance liquid chromatography (HPLC) (Zimba and Gitelson, 2006). Surface blooms might be identified by mapping Chl-a (Isenstein *et al*., 2014; Kutser, 2004; Moses *et al*., 2012) or PC (Hunter *et al*., 2010; Shi *et al.*, 2015) concentrations.

This study presents the first results of an Italian research project called BLASCO (Blending LAboratory and Satellite techniques for detecting CyanObacteria) for monitoring cyanobacteria in lakes based on EO data, and for tracking their blooms. The first section describes the cyanobacterial blooms occurring in Italian lakes in recent years. The second section concerns the contribution of remote sensing to cyanobacterial bloom mapping in four lakes. Satellite data obtained from Landsat-8 (L8) were used to assess the spatial distribution of scum and Chl-a concentrations during surface bloom events. The maximum peak-height (MPH) index (Matthews *et al*., 2012) was obtained from a 2003-2011 MERIS time-series to identify cyanobacterial surface blooms in meso-eutrophic subalpine lakes. Chl-a concentration products obtained for a shallow, turbid hypereutrophic lake were used to identify areas where the strongest blooms were likely to occur, also depending on the morphometric features of the lake basin as this might support an *in-situ* sampling strategy (Kiefer *et al*., 2015).

CYANOBACTERIAL BLOOMS IN ITALIAN LAKES

Toxic cyanobacteria are causing ecological and toxicological problems in Italy. Cyanobacterial blooms have been reported in 71 bodies of water (natural lakes and artificial reservoirs), and this figure probably underestimates the real situation. These events are linked to a general increase in the trophic status of the country's inland waters (Garibaldi *et al*., 1997, 2003; Carollo and Libera, 1992; Cordella and Salmaso, 1992).

Toxic blooms of freshwater cyanobacteria involve several filamentous genera, such as *Aphanizomenon* (Bruno *et al*., 1989), *Chrysosporum* (ex *Aphanizomenon*) (Messineo *et al*., 2009), *Cylindrospermopsis* (Manti *et al*., 2005) *Dolychospermum* (ex *Anabaena*) (Bruno *et al.*, 1994) and *Planktothrix* (Pomati *et al*., 2000; Messineo *et al*., 2006), as well as unicellular, colonial *taxa*, such as *Microcystis* (Bruno *et al.*, 1989), in which toxin production has been detected in specific populations.

Two species reportedly most often responsible for - *Planktothrix rubescens* (De Candolle ex Gomont) Anagnostidis and Komárek, and *Microcystis aeruginosa* (Kützing) Kützing (Messineo *et al*., 2006; Salmaso and Mosello, 2010) - both of them produce microcystins (Briand *et al*., 2003), a very common class of cyanotoxins, implicated in human and animal poisoning. *P. rubescens* typically inhabits deep lakes with a stable stratification and a metalimnetic layer in summer where this species adapted to low light and low temperatures can find the ideal growing conditions, as the phycoerythrin pigment gives rise to extremely effective light-capturing mechanisms (Steinberg and Hartmann, 1988), allowing its survival at lower depths than most algae (Davis *et al*., 2003). Many deep lakes and reservoirs in Europe are suitable for *P. rubescens* (Guiry and Guiry, 2011). In Italy, *P. rubescens* blooms have been reported in: Lakes Garda (Salmaso, 2000), Iseo (Garibaldi *et al*., 2003), Maggiore (Morabito *et al*., 2002), Orta (Morabito, 2001), Spino (Viaggiu *et al*., 2003) and Pusiano (Legnani *et al*., 2005) in the northern subalpine region; Lakes Albano and Fiastrone (Viaggiu *et al*., 2003), Nemi (Margaritora *et al*., 2005) and Vico (Manganelli *et al*., 2010) in Central Italy; and Lake Arancio (Naselli-Flores and Barone, 2007) in the South. In some cases, *P. rubescens* has been repeatedly reported as the dominant cyanobacterium in long-lasting bloom events (Viaggiu *et al*., 2004). The physiological mechanism behind *P. rubescens* blooms has been studied extensively in Lake Zurich (Walsby, 2005; Walsby *et al.*, 2006): the buoyancy of the filaments is regulated by the balance between carbohydrates production and consumption mediated by the underwater light and controlled by the depth of the mixed layer.

M. aeruginosa is a typical inhabitant of epilimnetic waters, adapted to high light conditions. This species is very common in Italy. In the north, its presence and/or blooms have been reported for Lakes Garda, Iseo, Maggiore, Caldonazzo, Canzolino, Serraia, Pusiano, Como and Monate (Manganelli *et al*., 2014; ISTISAN 35/11). In central and southern Italy, it has been detected in 6 lakes: Massacciuccoli, Trasimeno, Polverina, Castreccioni, Liscione and Cecita. It has also been found in 13 lakes in Sardinia and 3 lakes in Sicily (Manganelli *et al*., 2014; ISTISAN 35/11). Studies on the Sicilian reservoirs found blooms of *Microcystis* spp. associated with variations in water level, occurring common occurrence due to the Mediterranean climate (rainy winters and dry summers) and the island's river network (mainly consisting of temporary streams). In summer, water is drawn from lakes for irrigation and drinking purposes, causing a rapid drop in their le vel, that often prompts a lowering thermocline and disrupted stratification. The consequent marked change of mixing regime can mobilize the nutrients stored in the hypolimnion, boosting *Microcystis* blooms

(Naselli-Flores, 2003, 2014; Naselli-Flores and Barone, 2003, 2005, 2007). Its strong buoyancy also enables *M. aeruginosa* to counteract occasional mixing of surface waters (Salmaso *et al*., 2014b; ISTISAN 11/14).

CASE STUDIES: EXPLOITING REMOTE SENSING DATA

Three case studies were conducted in Lombardy (northern Italy), a region rich in both deep, medium-to-large, and small shallow lakes (Fig. 1). Materials and methods used for the three study cases analyzed are summarized in Tab. 1. Details for each study case are provided in the specific paragraphs.

Landsat-8 OLI for detecting blooms in Como and Pusiano lakes

The Como and Pusiano lakes (Fig. 1) are on the edge of the Landsat-8 OLI (L8) acquisition path, so they can be monitored on an 8-day (instead of the standard 16-day) cycle, which improves the chances of cyanobacterial blooms being identified because they sometimes last only a few days (O'Neil *et al*., 2012). Their spatial mapping with L8 imagery used in this study shows that the satel-

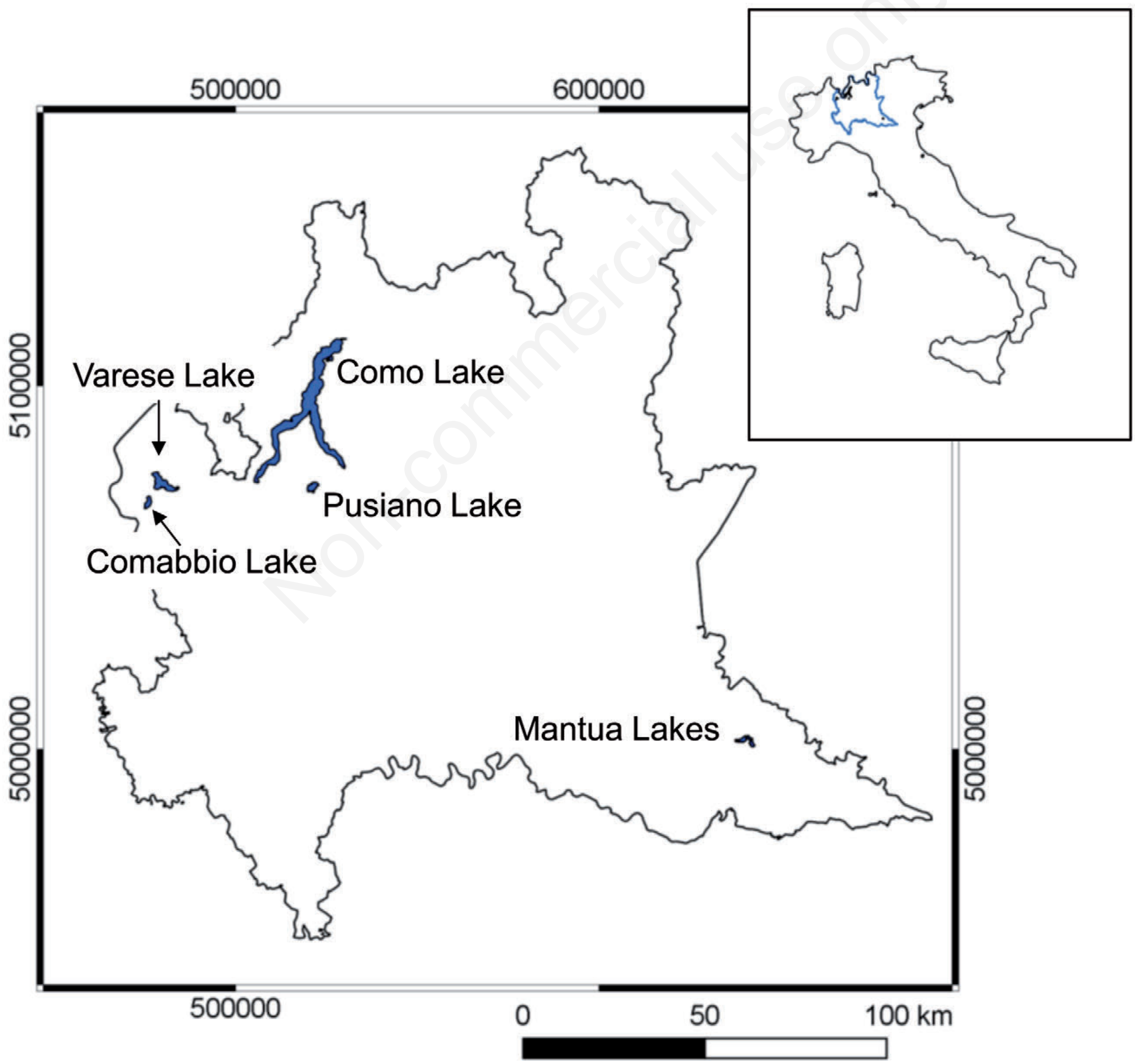


Fig. 1. Map showing the lakes in Lombardy Region (northern Italy) investigated in this study.

lite's 30-m pixel resolution suffices to capture the patchy distribution of cyanobacteria blooms (Fig. 2).

In early August 2013, an anomalous cyanobacterial bloom occurred in Lake Como (a large, deep lake in an oligo-mesotrophic state) that made its waters unsuitable for bathing or drinking for several days. The bloom was caused by *Dolichospermum lemmermannii* (Richter) Wacklin, Hoffmann and Komárek, a cyanobacterial species that produces surface scum. The bloom may be patchy, so L8 images were coupled with *in situ* monitoring, which was particularly challenging given the lake's size (145 km^2) and the bloom's short duration. Five L8 images acquired between the end of July and mid-August 2013 were radiometrically adjusted for water applications (Pahlevan *et al*., 2014), then atmospherically corrected with the 6SV code (Vermote *et al*., 2006). Surface bloom was detected using a band-ratio approach developed for similar purposes (Mayo *et al.*, 1995; Mahasandana *et al.*, 2001). Pixels where all three of the band-ratios i) b3 (561 nm) / b2 (483 nm); ii) b5 (865 nm) / b4 (655 nm); and iii) b3 (561 nm) / b4 (655 nm) higher than 1 were identified as scum. Using this method, numerous pixels revealed scum on the L8 image acquired on 1 August 2013, and none on or subsequent images. Fig. 2 shows the patchy distribution of *D. lemmermannii* at sites distributed all over the lake (total area = 431.8 ha). The satellite map was comparable with *in-situ* measurements obtained a day later, when the surface cyanobacteria concentration at the site in Fig. 2 was 365x10^6 cell L^{-1}. The scum was only mapped on 1 August 2013. It probably appeared as a result of significant rainfall blooming few days earlier. On 29 July 2013 precipitation occurred on Lake Como, 26 mm and 40 mm of 24 h cumulated precipitation, was recorded respectively in Como (south of the lake) and in Gera Lario (north) by ARPA Lombardia stations. A recent investigation (Callieri *et al*., 2014) found that *D. lemmermannii* blooms occasionally recorded in deep subalpine lakes in Italy were supported by nutrient pulses deriving from the mineralization of organic matter deposited along the lakeshore and released by rainfall event. Nutrients arriving from the lake's catchment area can stimulate phytoplankton growth, especially in oligo-mesotrophic lakes (Morabito *et al*., 2012), and combined with a seasonal increase in water temperature this would facilitate *D. lemmermannii* proliferation (Olrik *et al*., 2012; Salmaso *et al*., 2015).

L8 data acquired on 11 November 2015 captured a

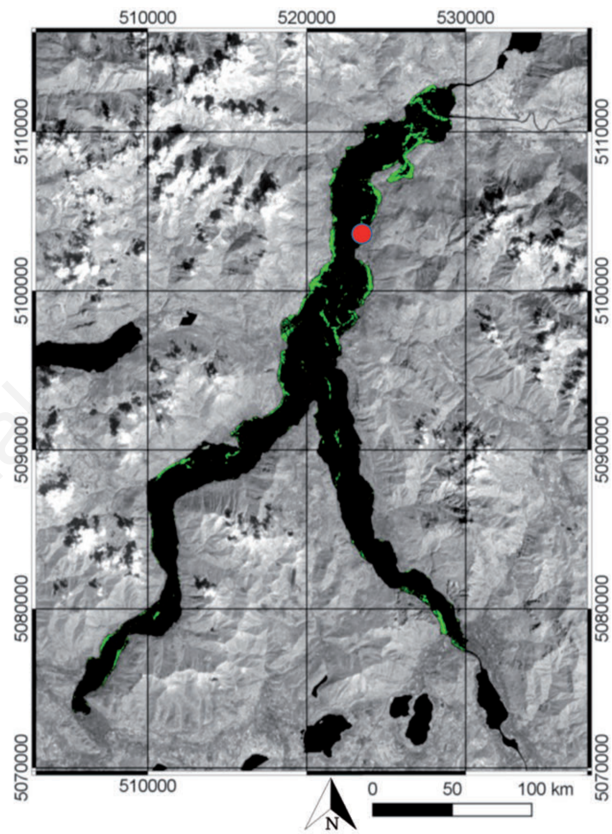


Fig. 2. Map of cyanobacteria scum (in green) in Lake Como from L8 data on 1 August 2013. The red circle marks the site of *in situ* measurements.

Tab. 1. Details on the sensors, number of images, approaches used and products derived for each lake object of this study.

Lake	Sensor	Number of images	Approach	Aim
Como	Landsat 8 - OLI	5	Band ratio	Cyanobacteria blooms identification
Pusiano	Landsat 8 - OLI	1	Bio-optical modeling	Maps of Chl-a concentrations
Comabbio	ENVISAT MERIS-FR	217	MPH	Multi temporal Cyanobacteria occurrence
Pusiano	ENVISAT MERIS-FR	248	MPH	Multi temporal Cyanobacteria occurrence
Varese	ENVISAT MERIS-FR	276	MPH	Multi temporal Cyanobacteria occurrence
Mantua	Hyperspectral (MIVIS-APEX-CHRIS)	7	Semi-empirical algorithms	Maps with zones characterized by high mean Chl-a concentration and high variability

cyanobacterial bloom in Lake Pusiano (a small lake south of Lake Como, in between its two branches). This lake has been hypereutrophic since the 1970s. Its total phosphorus concentrations have been gradually reduced by means of a water treatment plant, although cyanobacterial blooms have been observed again in recent years (Margaritora *et al*., 2006). During the L8 acquisition (was at 10:10 UTC) some stations were visited between 9:00 to 11:00 UTC to collect water samples and measure water reflectance spectra with a WISP-3 (Hommersom *et al*., 2012). The average Secchi disk depth was 1 meter (±20 cm), and no scum was apparent, while the cyanobacterial biomass in the uppermost layers of the water column was observed. The phytoplankton samples analyzed under the inverted microscope (400x magnification) according to Utermöhl (1958) revealed mainly *Woronichinia naegeliana* (Unger) Elenkin, but also *Dolichospermum* (cf. *planctonicum*) (Brunnthaler) Wacklin, L.Hoffmann and Komárek, and *M. aeruginosa*.

Chl-a concentrations were measured by spectrophotometry according to Lorenzen (1967) and HPLC. Photosynthetic pigments for HPLC analysis were extracted in 90% acetone, overnight in the dark, under nitrogen. The extract obtained was used to quantify Chl and its derivatives (in Chl derivatives units, CD) and total carotenoids by spectrophotometry. Individual carotenoids were detected by revers-phase HPLC with an Ultimate 3000 (Thermo Scientific). Specific pigments were identified by ion pairing, reverse-phase HPLC described in Guilizzoni (2011). PC concentrations were quantified with the spectrophotometer (SAFAS UVmc2) in 1 cm path-length cuvettes using the equations of Bennett and Bogorad (1973).

The average Chl-a concentration measured at the pelagic stations with no accumulated surface cyanobacteria (dots in Fig. 3) was 12 mgm^{-3} (±5 mgm^{-3}), while it was significantly higher at the two coastal stations and the other pelagic station where scum was found, at 173 mg m^{-3}, 550 mg m^{-3} and 97.4 mg m^{-3} respectively. The corresponding PC concentration were 490 mg m^{-3} and 5210 mg m^{-3} respectively for two coastal stations. HPLC on two surface samples collected in the pelagic zone revealed high concentrations of two cyanobacteria marker pigments (echinenone and myxoxanthophyll, with mean values of 14.7 mg m^{-3} and 16.4 mg m^{-3}, respectively). L8 data were radiometrically and atmospherically corrected using the same procedure as for Lake Como to compute the Chl-a concentrations and test the ability of L8 to capture blooms. The water reflectances obtained in the first four L8 bands were comparable with the spectra obtained *in situ* (with correlation coefficients of 0.57, 0.72, 0.83 and 0.79 for bands 1, 2, 3 and 4; ***P<0.001 for all four bands). L8-derived water reflectances were converted into Chl-a concentrations by adopting a spectral inversion procedure based on a bio-optical model (Giardino *et al*., 2012, 2014) parameterized with specific inherent optical properties of eutrophic water. For the three pelagic stations, the average Chl-a concentration was 10.7 mg m^{-3} (±1.4). The coastal area was more difficult to assess because the L8 band setting might be too coarse for the very high concentrations involved, but the Chl-a concentrations for the two coastal stations exceeded 30 mg m^{-3} (much higher than at the pelagic stations), consistently with field observations.

As in other inland water ecosystems, the patchy distribution of cyanobacterial blooms seen in Lake Pusiano was due mainly to wind (Webster and Hutchinson, 1994; Zilius *et al*., 2014; Wu *et al.*, 2015).

MERIS for monitoring cyanobacterial blooms in meso-eutrophic subalpine lakes

Small lakes south of the Alps are shallow, highly eutrophic, with highly variable Chl-a concentrations. Lake Varese is calcareous of glacial origin, sited to the west of Lake Maggiore. It has a mean depth of 11 m, and a surface area of 14.8 km^2. It is dimictic, with a summer stratification from May to November and an inverse stratification in winter. Lake Comabbio was originally linked to Lake Varese. It is polymictic, with a summer stratification from April to October. It has a mean depth of 4.6 m and a surface of 3.6 km^2.

For lakes Comabbio, Pusiano and Varese, respectively, 217, 248, and 276 MERIS Full Resolution (FR) images obtained from June to November (2003-2011) were processed to assess cyanobacterial blooms.

The MERIS FR Coast-Color level-1b images were pre-processed to correct the Rayleigh effect with the

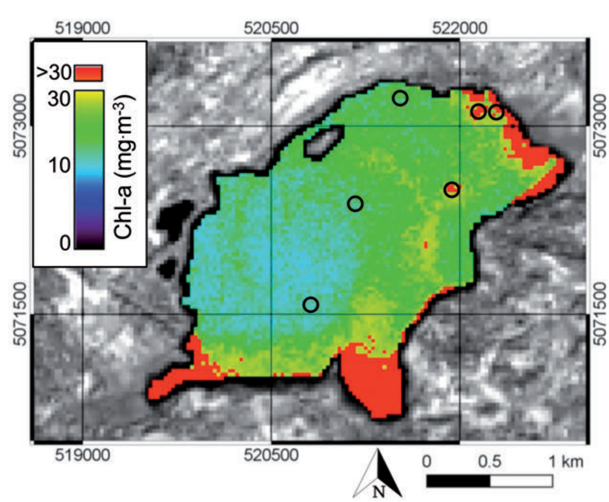


Fig. 3. Chl-a concentrations mapped in Lake Pusiano from L8 on 11 November 2015. The circles indicate the sites of *in situ* stations.

BEAM BRR (Bottom-of-Rayleigh Reflectance) processor. The product was then processed with Maximum Peak-Height (MPH) processor (Matthews *et al*., 2012). MPH exploits the BRR peaks in the red and near-infrared bands above a given baseline, which moves depending on the pigment concentrations. It provides a MPH index that is useful for calculating Chl-a concentrations, as towelled as flags for floating material and for eukaryote or cyanobacteria dominance for each pixel. Cyanobacteria dominance was estimated at 25%, 6%, and 12% on the images of the Comabbio, Pusiano and Varese lakes, respectively. The timing of this phenomenon varied from lake to lake, from season to season, and from to year. It was recorded most frequently in 2008 for Lakes Pusiano and Varese, and in 2011 for Lake Comabbio (Fig. 4). Considering the whole period, October was the month most frequently involved for Lake Comabbio, November for Lakes Pusiano and Varese (Fig. 5). Some of the events have been fully documented, *i.e.*, *P. rubescens* blooms in Lake Pusiano in Autumn 2010 (Salmaso *et al*., 2014a) and in Lake Varese in November 2011.

These results clearly show that cyanobacterial blooms

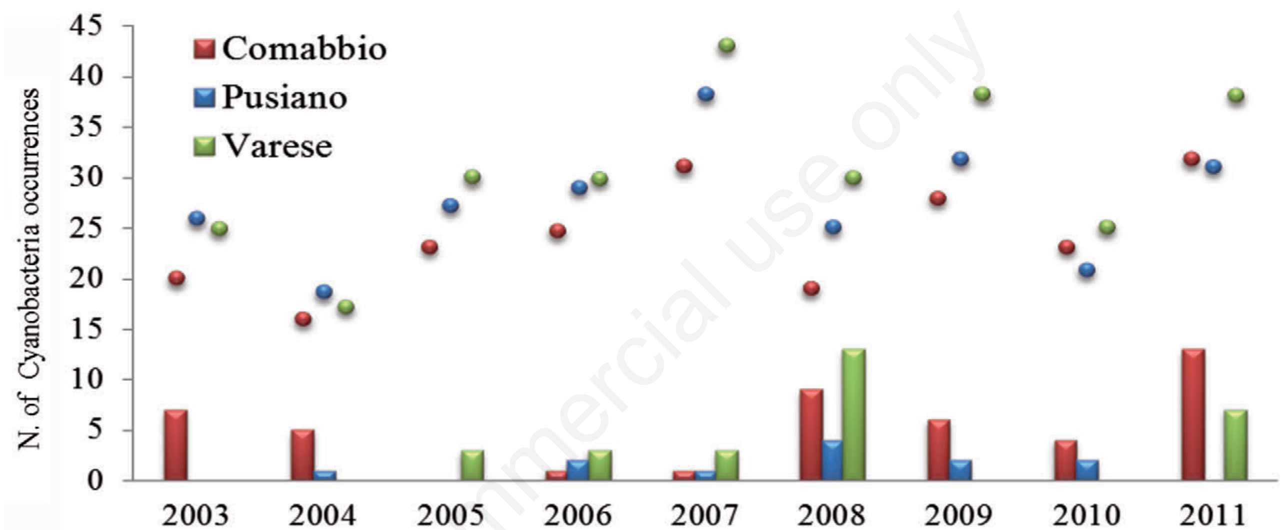
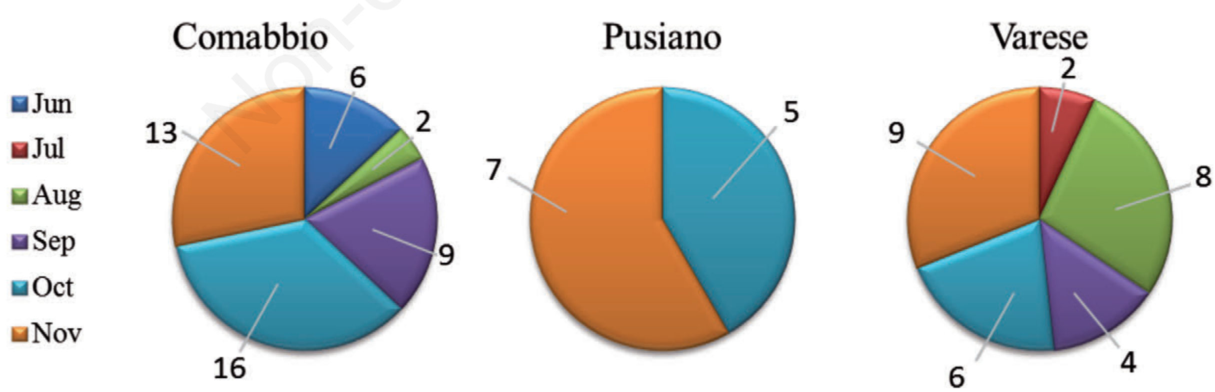


Fig. 4. Number of cyanobacterial blooms thereby year meso-eutrophic subalpine lakes.



Number of cloud-free images per month						
LAKE	Jun	Jul	Aug	Sep	Oct	Nov
COMABBIO	30	51	42	38	28	28
PUSIANO	37	57	53	40	32	29
VARESE	43	67	57	47	33	29

Fig. 5. Number of cyanobacterial blooms by month in the three meso-eutrophic subalpine lakes. In the table the number of cloud-free images per month.

in the subalpine lakes can occur even outside the bathing season, meaning that environmental agencies' typical monitoring programs can underestimate them if sampling is only done in summer. Recreational uses of these lakes often continue into late summer and early autumn, however, carrying a risk of intoxication for people and animals.

Imaging spectrometry for spatial analysis of Chl-a in hypertrophic waters

Lake Superior is the largest of three shallow hypertrophic lakes surrounding the town of Mantua, in northern Italy, with a surface area of 3.67 km^2 and an average depth 3.6 m. It is part of an artificial fluvial lake system created by damming the Mincio River in the 12^{th} century. Water levels in Lake Superior are regulated by the Vasarone dam and Vasarina gate (built in 2015), to ensure a constant 17.5 m asl (Pinardi *et al.*, 2011**,** 2015). Considerable nutrient loads enter the lake from its main tributary (the Mincio River), sustaining a dense phytoplankton community, with recurrent blooms that bring Chl-a concentrations up to about 100 mg m^{-3} (Bolpagni *et al*., 2014).

Seven images of Lake Superior were used to examine the spatial variability of mean Chl-a concentrations over time. All images were acquired from June to September by hyperspectral sensors on airborne platforms, *i.e.* MIVIS (2007-07-26) and APEX (2011-09-21 and 2014-09-27), and the satellite platform Proba-1 CHRIS (2008-06-29, 2008-09-16, 2011-08-28 and 2012-08-06). Chl-a concentrations were measured using the procedures described in Pinardi *et al*. (2015), during the season most

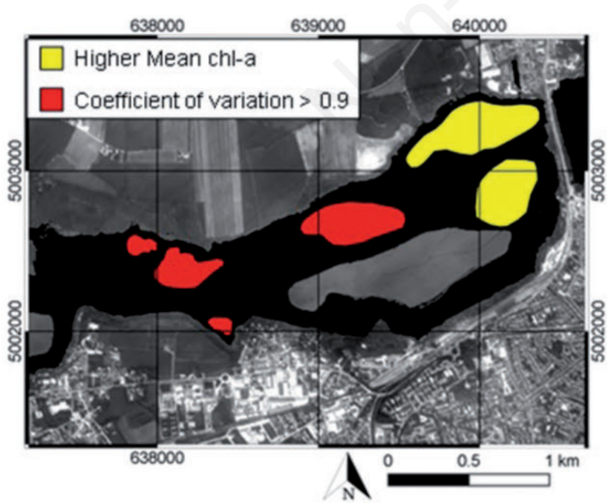


Fig. 6. Spatial analysis of Chl-a in Mantua Lake Superior obtained from hyperspectral images. The red zones had a higher coefficient of variation. The yellow zones had higher mean Chl-a concentrations, and would be appropriate for new *in situ* sampling sites.

associated with phytoplankton blooms, when local authorities, monitor the situation in accordance with the Italian Water Framework Directive guidelines.

The temporal analysis of the seven images was done in a GIS environment, using the same reference system (WGS84) and cartographic projection (UTM Zone 32N). Pixels were aggregated to a 15-m spatial resolution, which is the minimum common pixel size providing the spatially most detailed outcome. The GRASS function (r.series) and raster calculator tool were used to obtain statistics (*e.g.*, mean, standard deviation, coefficient of variation) of the temporal series for each pixel of Mantua Lake Superior. Some zones with particular statistical properties were selected and polygonized (Fig. 6): the yellow polygons identify areas with higher mean Chl-a concentrations (35-45 mg m^{-3}), and relatively lower standard deviations (less than 30 mgm^{-3}; mean coefficient of variation 0.75); the red polygons are areas with a high coefficient of variation (>0.9). The main hydrodynamic events influencing Chl-a distribution related to the combined effects of wind force and riverine current. The red zones identify areas where significant water circulation influenced the Chl-a concentrations, the yellow zones indicate areas where water stagnation favored phytoplankton bloom and accumulation.

CONCLUSIONS

This study aimed to describe the capabilities of remote sensing for mapping cyanobacterial blooms and to highlight the main advantages of such techniques, *i.e.* a synoptic view and frequent acquisitions to track dynamic phenomena. The case studies show that combining remote sensing with *in situ* measurements can help monitor cyanobacterial blooms in Italian lakes. Landsat-8 OLI data provided both the spatial distribution of cyanobacterial blooms in a small eutrophic lake, and the patchy distribution of scum in a large deep subalpine basin. The 10-year-long database of MERIS images enabled a dynamic mapping of cyanobacterial blooms affecting water quality in three small meso-eutrophic lakes, showing that algal blooms occurred for about 5 days a year, typically in late summer and early autumn. Air- and space-borne hyperspectral sensors were tested as a source of data for mapping Chl-a concentrations in Mantua's lakes, revealing that some zones of these lakes have higher Chl-a concentrations due to water circulation, suggesting the need to align *in situ* monitoring programs with the findings on hyperspectral images.

Future research will focus on further developing algorithms to enhance cyanobacterial mapping accuracy by including of semi-empirical and physically-based approaches to secondary pigments in cyanobacteria. The algorithms will also be adapted to new generation satellite sensors like the ESA Sentinel of the EC Copernicus program to access their fully operational EO capacity and improved spatial, radiometric and temporal resolutions. Sentinel-1 is a radar (SAR) instrument that can support scum detection even under cloud, and Sentinel-2 (like Landsat-8) can shed light on water quality. Most importantly, Sentinel-3 will be the successor of MERIS, with many optical bands specifically geared to water quality applications, and will assure continuous data acquisition for the next decades.

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