


Mapping phytoplankton blooms in deep subalpine lakes from Sentinel-2A and Landsat-8

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Abstract For effective lakes' management, high-frequency water quality data on a synoptic scale are essential. The aim of this study is to test the suitability of the latest generation of satellite sensors to provide information on lake water quality parameters for the five largest Italian subalpine lakes. In situ data of phytoplankton composition, chlorophyll-a (chl-a) concentration and water reflectance were used in synergy with satellite observations to map some algal blooms in 2016. Chl-a concentration maps were

derived from satellite data by applying a bio-optical model to satellite data, previously corrected for atmospheric effects. Results were compared with in situ data, showing good agreement. The shape and magnitude of water reflectance from different satellite data were consistent. Output chl-a concentration maps, show the distribution within each lake during blooming events, suggesting a synoptic view is required for these events monitoring. Maps show the dynamic of bloom events with concentration increasing from 2 up to 7 mg m⁻³ and dropping again to initial value in less than 20 days. Latest generation sensors were shown to be valuable tools for lakes monitoring, thanks to frequent, free of charge data availability over long time periods.

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Introduction

Phytoplankton blooms in lakes are prone to a high degree of change in space and time. In particular, cyanobacteria blooms are often characterized by complex dynamics in the vertical layers, when the taxa involved are capable of rapid vertical migration (Walsby et al., 1997). The typical dynamics of cyanobacterial blooms, which also have a very fast replication rates, make it difficult to perform a

quantitative monitoring of number of cells and spatio-temporal distribution as surface blooms can appear and disappear quickly, often within few hours (Sellner et al., 2003; Agha et al., 2012). The cyanobacteria principally responsible for forming blooms are mainly gas-vacuolate species. They are motile or buoyant and on occasions accumulate at the water surface to form a scum (Walsby & Reynolds, 1980). Surface blooms of cyanobacteria are strongly affected by environmental forces such as wind, temperature and sunlight; within few days a massive bloom can appear and completely disappear from the surface (e.g. Paerl, 1996; Wetzel, 2001; Hu et al., 2010). Surface blooms' spatial distribution is very depending on lakes' hydrodynamics, which tends to locate higher concentrations of cells in coastal and littoral areas (e.g. Vincent et al., 2009), coupled with calm conditions and reduced turbulence that allow buoyant migration to the water surface. Surface blooms can appear rapidly as result of an existing dispersed population from the upward migration (Reynolds, 1971). Short-term, periodic surface blooms can occur because of responses to daily meteorological events or cyclical changes in cell density. Under calm conditions, surface blooms frequently occur in the early morning as the respiratory demands during the hours of darkness consume the carbohydrate, which acts as ballast against the upward lift provided by the gas vesicles. This explains why blooms tend to disappear in the afternoon and to reappear in the morning (Paerl & Ustach, 1982).

To plan possible measures for managing and protecting natural ecosystems affected by extensive cyanobacteria blooms, it is important to obtain timely and synoptic information (Bresciani et al., 2016). The last can in fact support traditional in situ samplings (Liu et al., 2003; Nausch et al., 2008) which might be insufficient for meeting the requirements also in terms of costs. Further, a key factor in deciding the value of the data to be sought is the choice of sampling sites. Sample sites selection should be tailored to meet the overall aims and objectives of monitoring programmes, considering that the occurrence of a bloom is a function of the environmental conditions and the resource requirements of the organisms (Chorus & Bartram, 1999). Within this context, synoptic, frequent, and global Earth Observation (EO) data might provide valuable data to improve the monitoring of phytoplankton blooms in freshwater ecosystems (Hestir et al., 2015). For the last decades, EO data have

been successfully applied for mapping blooms and phenology (Wang & Shi, 2008; Stumpf et al., 2012; Bresciani et al., 2014; Matthews & Odermatt, 2015). These studies were typically focused on the retrieval of chlorophyll-a concentration (chl-a), commonly used as a proxy of phytoplankton biomass. Chl-a was mapped already in 1974 (Strong, 1974) and it was the first parameter derived quantitatively from Ocean Color satellites (Matthews, 2017), a suite of satellite sensors, having the optimal resolutions (e.g. radiometry, daily revisiting time) for studying ocean phytoplankton. In particular, the ESA MERIS (MEdium Resolution Imaging Spectrometer) ocean colour sensor, with its spatial resolution of about 300 m and spectral bands suitable for ocean colour studies, used to calibrate the algorithms for detecting chl-a in clear and turbid waters, has been successfully used for studying lakes from 2002 to 2012.

The latest generation of medium resolution multi-spectral sensors on board of Landsat-8 (L8) and Sentinel-2 (S2) satellites, is now offering advanced opportunities for synoptic, fine-scale, and high-frequency monitoring applications in lakes. Differently from ocean colour satellites, they have not been specifically designed for observing water but they are both promising for detailed water quality analysis (Kutser, 2004; Pahlevan et al., 2014) due to (i) radiometric sensitivity (> 12-bit quantization) (Hedley et al., 2012; Dörnhöfer & Oppelt, 2016); (ii) spatial resolution of 10–30 m; (iii) frequency of overpass (up to every 2–3 days combining L8 and S2 satellites); and (iv) the improved spectral band configuration in the visible–near-infrared wavelengths range.

L8 was launched in 2013 and carries on board the Operational Land Imager (OLI) sensor. It is operating with a spatial resolution of 30 m at ground, acquiring images every 16 days (Irons et al., 2012; Roy et al., 2014), that might be reduced to 8–9 days in case target areas lie in two adjacent orbits (Brando et al., 2015). L8 aims to provide data continuity to the global NASA Landsat Earth observation program that started in the 1970s. The OLI sensor has been evaluated to be suitable for the assessment of water quality and water constituents in many water bodies such as lakes, estuaries, rivers and coastal zones (Giardino et al., 2014; Vanhellemont & Ruddick, 2014; Brando et al., 2015; Franz et al., 2015; Lobo et al., 2015; Slonecker et al., 2016).

The Sentinel-2A and B satellites (S2A/B), the twin Copernicus ESA mission operating the MSI (MultiSpectral Instrument) sensors, are acquiring imagery every 5 days, with a spatial resolution from 60 to 10 m in the visible region (Fig. 1). S2A, launched on 23 June 2015, has been already used for the assessment of water constituents, for example, in the oligotrophic waters of Starnberg Lake (Dörnhöfer et al., 2016) and in the humic lakes (Toming et al., 2016). Moreover, the MSI bands close to the red-edge are promising for resolving productive turbid waters. Overall, S2A suggests great potential in lake ecology studies and in the retrieval of water quality indicators at fine spatial scale synoptically.

Combining L8 and S2A/B data, a multispectral global coverage with a pixel size of 10–30 m is available approximately every 3 days (Yan et al., 2016). S2B, launched on 7 March 2017, was not exploited in this study, which is focused on some 2016 events.

For accurate and consistent mapping, sensors inter-calibration as well as pre-processing methods remain a challenging issue (van der Werff & van der Meer, 2016). To accurately estimate water quality parameters and in particular chl-a concentrations as a proxy of

phytoplankton blooms, remotely sensed data require to be corrected for radiometric and atmospheric effects for estimating remote sensing reflectance (RRS). RRS is in fact one of the most used among radiometric quantities to estimate water quality parameters. Among the effects signal needs to be corrected for, the most relevant is due to the atmosphere, because water reflectance represents a small contribute (< 20%) to the total reflected signal sensed from space (Maul, 1985), while the most of the signal is due to interaction between electromagnetic signal and aerosol and gases particles. Several algorithms and techniques have been developed and used in the last decades based on radiative transfer equations or on their simplified versions (e.g. Vermote et al., 2006; Richter & Schläpfer, 2014; Moses et al., 2017). These methods might also include the correction of adjacency effects (Sterckx et al., 2015) or of sources of radiometric noise that might hinder the use of optical satellite data due to glint, clouds, clouds shadows and cirrus (e.g. Amin et al., 2013). If not corrected for, these effects have to be properly masked out before estimating chl-a concentration. Then, a variety of algorithms has been developed for estimating chl-a concentration from RRS (e.g. Odermatt et al., 2012).

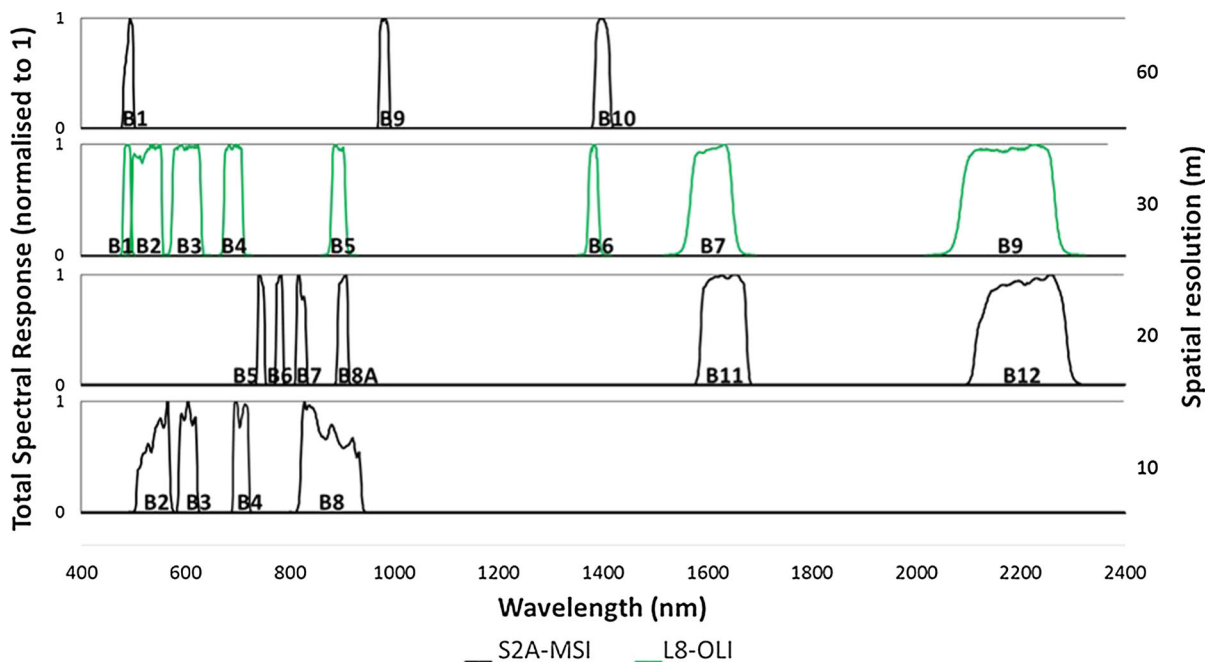


Fig. 1 Spectral response functions and spatial resolution for S2A-MSI and L8-OLI bands. B1 to B12 indicate band number for both sensors. L8-OLI panchromatic band 8 is not shown

The inferring methods make use of site-sensor-specific empirical/semi-empirical algorithms (e.g. Gitelson et al., 2008; Gilerson et al., 2010; Gurlin et al., 2011; Kudela et al., 2015), or spectral inversion schemes of physically based bio-optical models (e.g. Brando & Dekker, 2003; Giardino et al., 2007; Heege et al., 2014; Gege, 2014). Bio-optical models have the advantage of being sensor independent and applicable to every lake, presuming each lake-specific inherent optical properties (SIOPs) are known.

The aim of this work is to use the advantage in using new satellites generation optical sensors (i.e. L8 and S2A) to increase the information gathered from in situ data on algal blooms dynamics in deep subalpine lakes. The performance of a sensor-independent physically-based image-processing chain, calibrated with SIOPs of lakes, is evaluated by comparing the RRS and chl-a products obtained from L8 and S2A with match-ups of in situ data. The values added by remote sensing products in terms of frequency and synoptic observation could be an important integration to limnological measurements and hydrological modelling for lakes water monitoring and for ecological and management purposes. Toward this, chl-a concentrations derived from satellite images are analysed with respect to the quality classes defined by the EU Water Framework Directive (WFD, Directive EC 2000/60) for the selected lakes.

Materials and methods

Study site description

The lakes investigated in this study are located in Northern Italy, in the subalpine lakes district: from West to East, they are Maggiore, Como, Iseo, Idro and Garda (Fig. 2), the morphometric characteristics of these lakes are reported in Table 1. Their average depth ranges from 60 to 178 m and in total they store up to 126 km³ of water, which represents an invaluable resource. The lakes are in fact located in one of the most densely populated and highly productive area of the country, and represent an essential strategic water supply for agriculture, industry, fishing and drinking. Moreover, they are an important resource for recreation and tourism with their attractions of landscape, mild climate and water quality. The importance of the deep Southern subalpine lakes is shown by the

fact that, as early as the end of the nineteenth century, they were an object of study by a number of authors (for a review see Manca et al., 1992).

Because of their geographic position in the temperate belt, they should be warm monomictic basins; however, their great depth as well as specific morphological and climatic conditions cause them to be regarded as potentially oligomictic. In fact, the complete winter overturn can only occur during particularly cold and windy winters and is therefore a rather uncommon event (Ambrosetti & Barbanti, 1992). An exception is Lake Idro, which is in a permanent meromictic situation.

All these lakes underwent an eutrophication process during the 1960s. However, their present trophic conditions are now different, due to the variable degree of eutrophication reached and, except for Lake Idro, to the extent of recovery due to the variable timing and severity of the measures adopted against eutrophication.

Lake Maggiore, oligotrophic by nature, as shown by early limnological studies (Baldi, 1949) and by analysis of the sedimentary pigments (Guilizzoni et al., 1983), reached a trophic state close to eutrophy at mid-seventies, when the P loads peaked and the maximum in-lake TP concentration was recorded (around 30 µg l⁻¹ at winter mixing; Ruggiu et al., 1998). Since that time, the P loads have been gradually reduced by various means, including the establishment of sewage treatment plants and the reduction of total phosphorus in detergents. As a result, the TP values have gradually decreased to some 10 µg l⁻¹ (Ruggiu et al., 1998).

Lake Como reached a peak of TP at the end of the 1970s, with a maximum concentration around 80 µg l⁻¹ at the spring overturn (Ambrosetti, 1983). From 1986, the reduction of the nutrient load, following an abatement of phosphorus in detergents, a decrease in industrial effluents and the commissioning of new sewage treatment plants, caused the TP concentration in the Como basin to fall from 71 µg l⁻¹ (Mosello et al., 1991) to 40 µg l⁻¹ at the spring overturn in 1992. More recent data (Buzzi, 2002), indicate a TP concentration at overturn close to 30 µg l⁻¹.

Lake Garda never reached TP concentration as high as the other lakes: its trophic condition always remained between oligotrophy and mesotrophy. Peaks of TP around 20 µg l⁻¹ were recorded in 1999 and

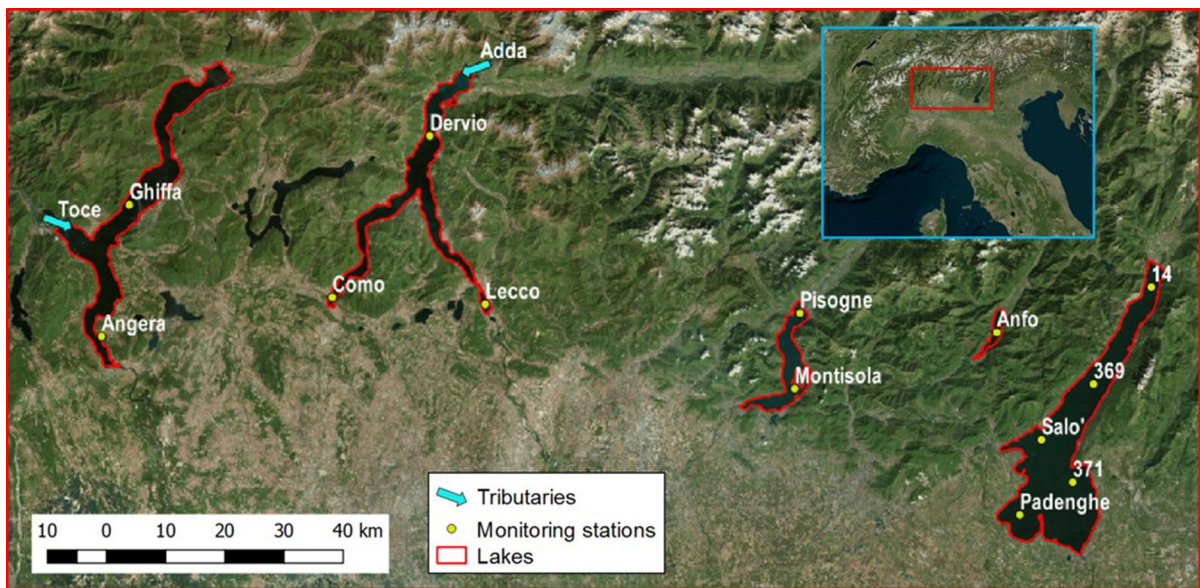


Fig. 2 The deep lakes investigated in the study: from West to East, Maggiore, Como, Iseo, Idro and Garda. The locations of the water quality monitoring stations handled by the regional

water authorities are indicated by yellow dots. The arrows show the river tributaries of Maggiore and Como mentioned in the study

Table 1 Main morphometric characteristics of the studied lakes

	Garda	Maggiore	Como	Iseo	Idro
Latitude N	45°36'	45°58'	46°00'	45°43'	45°46'
Longitude E	10°38'	8°39'	9°15'	10°05'	10°30'
Altitude (m a.s.l.)	65	193	198	186	368
Area (Km ²)	368	213	146	62	11.5
Maximum depth (m)	350	370	410	251	124
Mean depth (m)	136	178	154	123	59.5
Volume (Km ³)	50	37.5	22.5	7.6	8.5
Renewal (years)	26.8	4.1	4.5	4.1	0.99

2004, after deep mixing episodes (Salmaso et al., 2007). The situation is now stable, with a maximum TP concentration at overturn equal to $18 \mu\text{g l}^{-1}$ and an annual average equal to $13 \mu\text{g l}^{-1}$ (Salmaso et al., 2016).

Lake Idro is meromictic: this characteristic is, at least in part, due to the nature of the geological substrate, rich in gypsum, and to the position of the lake, surrounded by high mountains and sheltered from the winds. Moreover, the eutrophication of the lake strongly exacerbated the meromictic condition (Garibaldi et al., 1995). The upper mixed layer is 40 m deep at most. Concurrently, the deeper water layer accounts for $\sim 50\%$ of the total water column, which is anoxic (Nizzoli et al., 2012). The most recent

investigation (Osservatorio dei Laghi Lombardi, 2005) reports an epilimnetic TP value of $24 \mu\text{g l}^{-1}$ and a hypolimnetic concentration equal to $224 \mu\text{g l}^{-1}$.

Lake Iseo appears to be meso-eutrophic, although its phosphorus concentration showed an increasing trend along the year. Average values were almost stable in the decade 1975–1985, between 25 and $35 \mu\text{g l}^{-1}$, then started to increase, reaching the actual values, close to $60 \mu\text{g l}^{-1}$ (Mosello et al., 2010). The lake presents a deep anoxic layers, below 150 metres depth, due to the lack of a deep water circulation in the most recent period.

Concerning cyanobacteria, blooms have been recorded throughout the whole group of deep sub-alpine lakes (Salmaso et al., 1994; Garibaldi et al.,

2000, 2003; Salmaso, 2005; D’Alelio & Salmaso, 2011). The species involved are, mainly, *Dolichospermum lemmermannii* (Richter) Wacklin, Hoffmann & Komárek (2009) (in lakes Maggiore, Garda, Iseo and Como), *Microcystis aeruginosa* (Kützing) Kützing 1866 (Lake Como, Iseo and Idro) and *Planktothrix rubescens* (De Candolle ex Gomont) Anagnostidis & Komárek (Lake Idro and Iseo).

Over the last 30 years, algal blooms seem to have become a constant element in all the deep subalpine lakes, although it is interesting to point out the recent increase of the frequency of cyanobacterial blooms, even in those lakes, such as Maggiore and Garda, where these events were mostly unexpected, due to the low trophic status of these basins.

Image processing

The assessment of bloom events by using the chl-a concentration as a proxy, was developed from L8 and S2A imagery acquired in late summer 2016, which is the most likely season of cyanobacterial bloom events in subalpine deep lakes. In particular, a series of cloud-free images, 2 from L8 and 9 from S2A, respectively, acquired from August to September 2016, were used for mapping the blooms (Fig. 3, ‘Detection’ panel). Another set of imagery, 10 L8 images and 8 S2A images, spanning between 2015 and 2017, were instead used to evaluate the performance of the image-processing chain (Fig. 3, ‘Validation’ panel).

A common image-processing chain to obtain chl-a concentration maps was applied to both S2A and L8 Top Of Atmosphere (TOA) radiances products. In particular, S2A TOA radiances were computed from the S2A-MSI level2 TOA reflectance images (downloaded from Copernicus Open Access Hub) by using the SNAP tool *ReflectanceToRadianceOp*. For L8-OLI, (downloaded from U.S. Geological Survey *EarthExplorer* portal) the level1B products were converted into TOA radiances by applying radiometric calibration gains and then rescaled using aquatic-specific gains (Pahlevan et al., 2014). Both S2A and L8 TOA radiance products were atmospherically corrected through an algorithm based on the vector version (6SV) of *Second Simulation of the Satellite Signal in the Solar Spectrum* (Vermote et al., 2006). The code was parametrized with Aerosol Optical Thickness (AOT) and aerosol microphysical properties information at the time of imagery acquisition, collected from AERONET stations (<https://aeronet.gsfc.nasa.gov/>) located in the study area (i.e. Ispra and Sirmione_Museo_GC, on the coasts of Maggiore and Garda lakes, respectively). AERONET measurements are mandatory to define appropriate atmospheric profile and aerosol type used for atmospheric correction. When AERONET data were not available, AOT from MODIS-Terra Level-3 products were used (obtained through Giovanni portal <https://giovanni.sci.gsfc.nasa.gov/>) together with a standard percentage of aerosol composition for North Italian Po Valley (*Dust-like*: 40%, *Water-Soluble*: 44%, *Oceanic*: 5%, *Soot*:

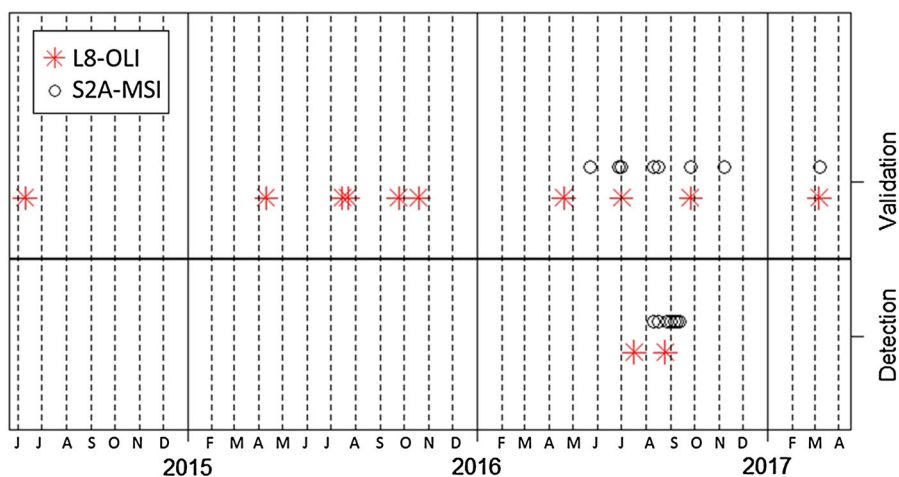


Fig. 3 Dates of images used for the validation of atmospheric correction and chl-a products and/or for the detection of cyanobacterial algal blooms

11%). The 6SV products were corrected for specular effect and converted in RRS according to Mobley (1999).

RRS products were then used as input of a bio-optical model for estimating the chl-a concentration called BOMBER (Bio-Optical Model-Based tool for Estimating water quality and bottom properties from Remote sensing images; Giardino et al., 2012). The bio-optical model BOMBER is a typical four-component model developed to determine water constituents in optically complex waters from a spectral inversion procedure. Giardino et al. (2012) provided a full description of the model. Relevant for this study is the parametrization of the model, which relies on SIOPs gathered in situ, as described in Giardino et al. (2014) and in Bresciani et al. (2016). In particular, the model performed the inversion by choosing the specific absorption spectra due to phytoplankton and chl-a concentration which provided the minimum error. During the optimization, the chl-a concentration was free to vary between 0.01 and 50 mg m⁻³, while two specific absorption spectra due to phytoplankton were used, one suitable for clear waters (Giardino et al., 2014) and the other for more productive waters (Bresciani et al., 2016).

RRS and chl-a products retrieved through this processing chain were initially compared to in situ data collected from both the monitoring activities of the water authorities (ARPA) and from dedicated radiometric campaigns, the latter including also in situ measurements of RRS, in addition to water samples gathered for subsequent laboratory analysis for determining the chl-a concentration. In order to evaluate the benefit from using EO products, mean chl-a concentration values were extracted by 3-by-3 pixel regions corresponding to in situ stations, where in situ measurements have been carried out. In these stations, coefficient of variation index was calculated and compared to the maximum obtained from corresponding images.

In situ measurements and validation

The results of the image-processing chain for 8 S2A and 10 L8 images, both in terms of atmospheric correction and bio-optical model inversion, were compared to in situ measurements of RRS and chl-a concentration retrieved during field campaigns

performed synchronous to satellite overpasses (or with a maximum of 1-day time difference).

In particular, 9 radiometric and limnologic field campaigns were conducted in Maggiore, Iseo and Garda lakes, measuring both RRS and chl-a concentration between 2014 and 2017. Radiometric measurements were performed with a WISP-3 (Hommersom et al., 2012), a spectroradiometer developed for gathering RRS spectra of water in the range of 400–800 nm, as well as with a full range of radiometers (i.e. a FieldSpec ASD-FR and a SpectralEvolution (SE)), operating between 350 and 2500 nm and operated according to Bresciani et al. (2013). In particular, WISP-3 simultaneously measures water and sky radiances at 40° at Nadir, and total irradiance, by three different optics and appropriate geometry. With ASD and SE, these measurements were sequentially acquired by mounting a remote cosine receptor and a 5° lens and keeping the angles consistent to the SeaWiFS protocol (Fargion & Mueller, 2000). Integrated water samples between the surface and the Secchi-Disk depth were collected by using a Van Dorn sampler. The water was then filtered in situ for subsequent laboratory analysis. Chl-a concentrations, extracted with acetone, were determined in laboratory via spectrophotometric method (Lorenzen, 1967). In addition, chl-a concentration time series data, collected from monitoring measurements by local authorities, were used. From these time series, synchronous measurements to satellites overpass were selected: 1 date on Lake Como and 1 date on Lake Garda.

Water samples for phytoplankton species and composition were also available for four (Maggiore, Como, Idro and Garda) of the five investigated lakes. The taxonomic composition and density of phytoplankton were analysed under a Zeiss inverted microscope according to Utermöhl's method (1958). Phytoplankton organisms were identified at the species level or, if not possible, assigned to a genus only. Lakes have been sampled once, during August, except Lake Maggiore, where two samples have been collected in September.

For both S2A and L8 sensors, where in situ measurements of RRS were available, validation of atmospheric correction was conducted comparing mean RRS extracted from 3-by-3 pixel regions centred over each in situ station. Root mean square error (RMSE) was calculated, for each band used later

through BOMBER, as in Eq. (1), where y_{insitu} e y_{EO} are, respectively, the measured in situ values and the RRS values extracted from Remote Sensing products, and n is the number of stations.

$$\text{RMSE} = \frac{\sqrt{\sum_{i=1}^n \left(\frac{y_{\text{insitu},i} - y_{\text{EO},i}}{y_{\text{insitu},i}} \right)^2}}{n} \quad (1)$$

The mean values of chl-a concentration extracted by 3-by-3 pixels region of image-derived products and corresponding to in situ stations were also compared to in situ measurements by computing the RMSE.

Results

Validation results

Good results were obtained for both RRS and chl-a concentration derived from S2A and L8, comparing them against in situ radiometric and limnologic measurements. RMSE for atmospheric correction products (Fig. 4) showed good agreement between in situ measured and estimated RRS. In particular, for the S2A images, the RMSE values in the visible region were between 0.002 and 0.003 sr^{-1} , with lower values in the near-infrared band. For L8, RMSE values were slightly higher (0.003 sr^{-1}) than S2A in blue regions, while in green and red L8 bands it was more efficient than S2A.

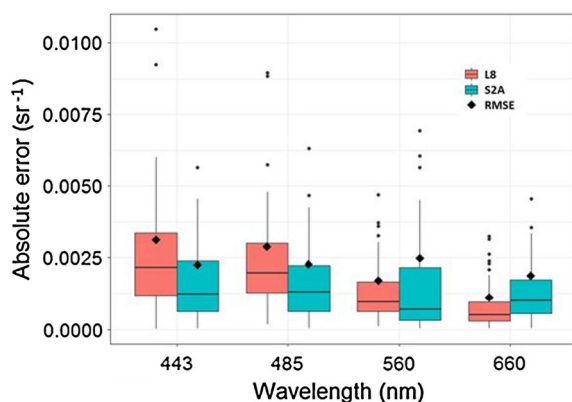


Fig. 4 Boxplot of absolute error and corresponding RMSE, obtained for each band for L8 and S2A images used for the validation analysis, from the 9 field campaigns for a total amount of 91 match-ups (60 for L8 and 31 for S2A). In the image, bands are indicated with the central wavelength of bands mediated between the two sensors

A further step in the validation is the optical closure, a plot where RRS spectra obtained from (1) in situ data; (2) atmospherically corrected images; and (3) forward bio-optical modelling are plotted aiming to obtain from all of these independent measurements a similar spectrum. Figure 5 shows this comparison for two cases in which the satellite-derived RRS spectra were obtained from S2A and a L8 images, respectively. In both cases, the forward modelling was obtained by simulating the RRS spectra based on in situ data input (in particular, the actual chl-a concentration and the more appropriate specific absorption phytoplankton coefficient).

The chl-a concentration products, estimated by applying the bio-optical model to the atmospherically corrected satellite L8 and S2A images, were also found in good agreement with in situ measured concentration (Fig. 6), within a range of chl-a concentration varying from 1 up to 7 mg m^{-3} .

Bloom maps and in situ analysis

Chl-a concentration maps in Fig. 7 show the bloom events detected in the 2016 for Garda, Maggiore, Como and Idro. In case of Lake Iseo, imagery analysis did not reveal any significant bloom at that time and this lake was therefore left out from the analysis. In Lake Garda, chl-a concentration was higher during the bloom, in particular in the Southern part of the lake, where on August 17th, it increased up to 8 mg m^{-3} . Even if concentration values do not reach values typical of eutrophic environments, we considered these events as blooms, according to Reynolds & Walsby (1975). Similar increase was present in the Northern part of Lake Como on August 10th (up to 10 mg m^{-3}): in this period, spatial pattern allowed also recognizing the bloom extension from the North, towards Southern part of the lake. Also in Lake Maggiore the increase, showed by September 9th image, was more evident in the area next to the Toce Western tributary (up to 7 mg m^{-3}). For Lake Idro, for which in situ data showed the presence of cyanobacteria since the first 10 days of August, imagery also clearly show the presence of bloom event on August 17th and 27th and September 6th by further revealing different spatial distributions.

Specifically, in Lake Idro the phytoplankton analysis of the sample of August 10th detected the

Fig. 5 In situ measurements, BOMBER simulation and Remote Sensing products of RRS, from L8 image of Lake Maggiore on 24/9/2015 (a) and from S2A image of Lake Iseo on 26/9/2016 (b)

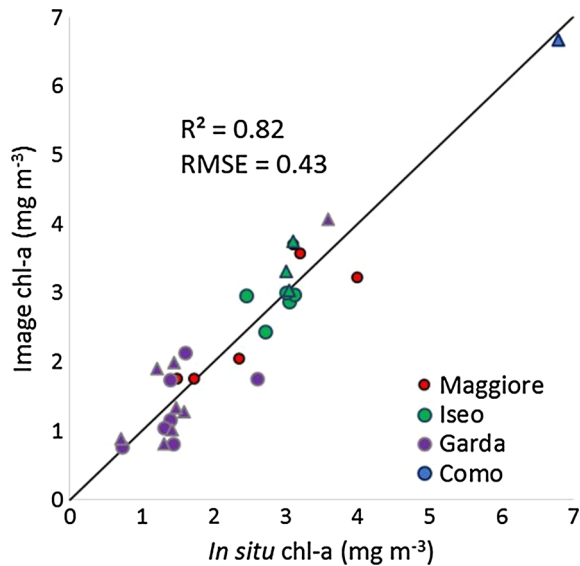
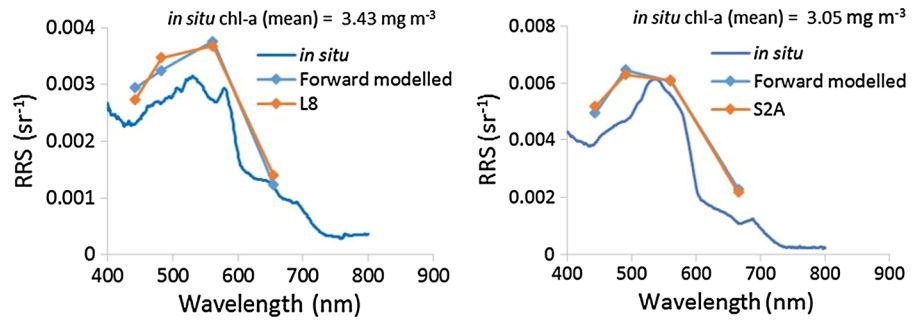


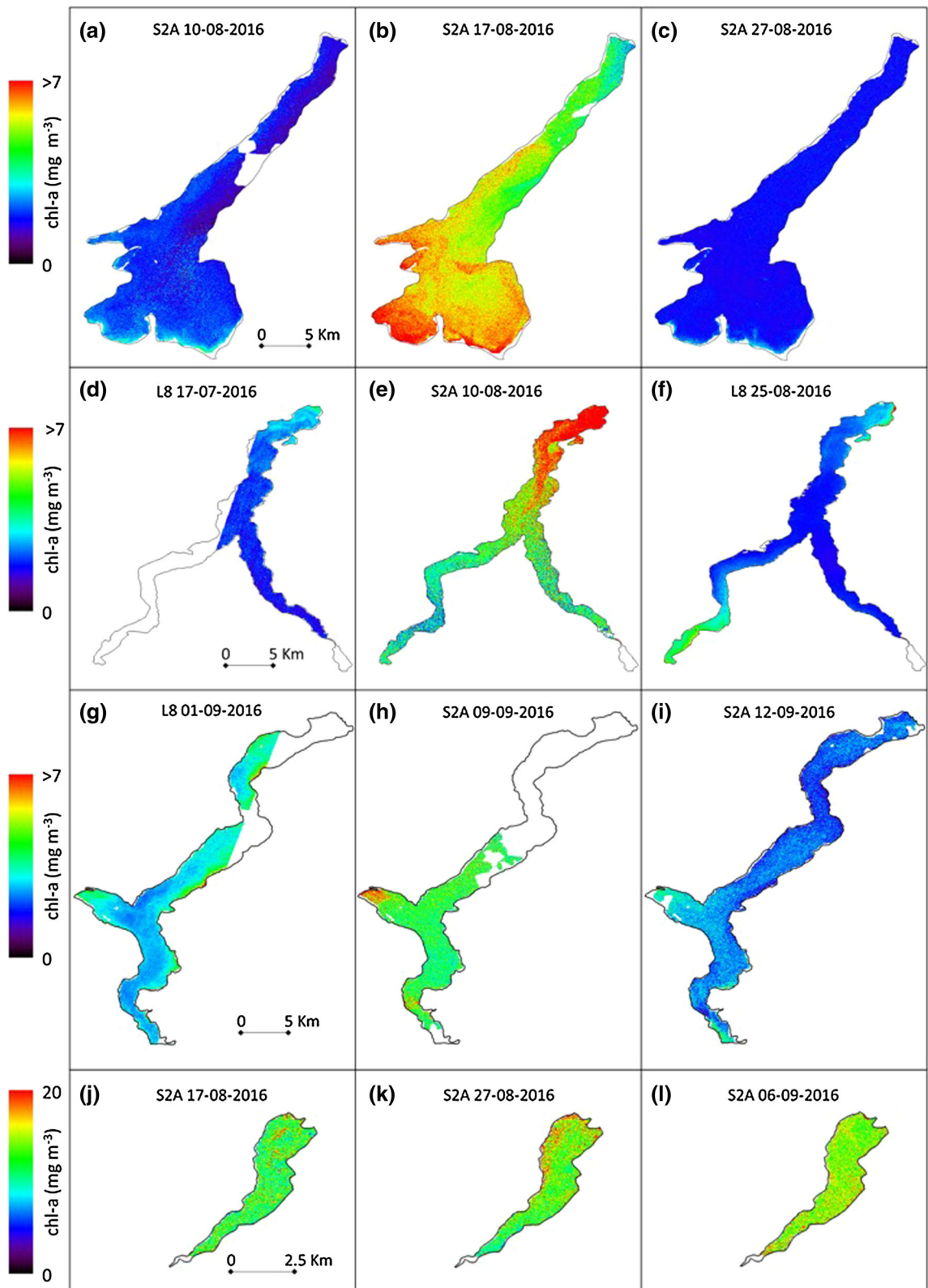
Fig. 6 Chl-a concentration measured in situ (x -axis) and estimated from remote sensing data (y -axis), RMSE and Coefficient of determination R^2 . Triangles and circles indicate, respectively, S2A and L8 products. Solid line is the bisector of the first quadrant

presence of cyanobacteria (40% of the total abundance), represented mainly by *Chroococcus* cf. *turgidus* (Kützing) Nägeli (27%). Similar importance has the chrysophytes, represented by the genus *Ochromonas Vysotskii* [Wysotzki, Wyssotzki] (27%), while the diatom *Fragilaria crotonensis* Kitton 1869 contributed only 8.7% in terms of abundance, although reaching 52.5% as biomass.

In Lake Garda, the phytoplankton sample of August 2016 revealed a *Dolichospermum lemmermannii* bloom. Different species of cyanobacteria, belonging to Chroococcales and Oscillatoriales, represented *Anathece clathrata* (W. West & G.S. West) Komárek, Kaštovský & Jezberová, comb. nov., *Aphanothece nidulans* P. Richter in Wittrock & Nordstedt 1884, and

Planktothrix rubescens, *Pseudanabaena limnetica* (Lemmermann) Komárek 1974, *Tychonema bourrellyi* (J.W.G. Lund) Anagnostidis & Komárek 1988, respectively, were found in the sample analysed for Lake Como on August 11th. However, in terms of biomass, *Planktothrix rubescens* (51.2%) and *Tychonema bourrellyi* (21%) are the dominant species.

Blooms of cyanobacteria were present in Lake Maggiore during September 2016. Water samples were collected in Lake Maggiore for phytoplankton analysis on September 10th, 19th and 26th. The first sample (September 10th) revealed the presence of the typical surface bloom of *D. lemmermannii* in the Southern part of the lake. *D. lemmermannii* contributed to almost the total of the bloom in terms of density (91.1%), while minor importance had *Microcystis aeruginosa* (5%) and *Dolichospermum planctonicum* (Brunnthaler) Wacklin, L. Hoffmann & Komárek 2009 (2.5%). In the central part of the Lake Maggiore, two specimens revealed the presence of several cyanobacteria mostly belonging to Oscillatoriales. The blooms were mainly due to *Pseudanabaena* spp., an unusual condition in Lake Maggiore, as this taxon commonly blooms in eutrophic water bodies (Mayer et al., 1997; Zwart et al., 2005); nonetheless, its characteristics are poorly investigated (Acinias et al., 2009) even widely distributed as it occurs in diverse aquatic as well as in benthic environments (Castenholz et al., 2001; Zwart et al., 2005; Diez et al., 2007). In the two samples collected in the central lake station, several species belonging to Oscillatoriales have contributed to the algal bloom, among which *Pseudanabaena* dominated, both in terms of density (68.6 and 57.0%, respectively) and biomass (47.2 and 52.6%). *Dolichospermum* genera (*D. lemmermannii* e *D. planctonicum*) contributed, respectively, to 27.9 and 7.6% in terms of biomass.



◀ **Fig. 7** Chl-a concentration maps for the time windows of algal bloom events on each lake. From the top: Garda (a–c), Como (d–f), Maggiore (g–i) and Idro (j–l)

WFD perspective

The Water Framework Directive (WFD, Ferreira et al., 2007) aims to establish a framework for the protection of inland surface waters, transitional waters, coastal waters and groundwater. In practice, this means that State Members have to achieve good ecological status (the scale is High, Good, Moderate, Poor and Bad status) or good potential in terms of chemical and ecological parameters within time limits set in the directive. Chl-a concentration is one of the key parameters used in the status classification.

In Fig. 8, the histograms of the chl-a concentration distribution for each image related to the WFD scale status are shown. The results show the advantages to use synoptic view characterizing the whole water surface of the single lakes. Results in Fig. 8 obtained from Lake Garda and Lake Maggiore for the images without the blooms are comparable to results obtained from MERIS time series analysis (GLaSS, 2015).

The multitemporal analysis clearly showed the worsening of the quality of lakes, during the algal bloom event, and demonstrates the usefulness of high-frequency monitoring tools, allowing the detection of phytoplankton events, which could potentially affect the final classification of the ecological status.

The detection and the analysis of the characteristics and spatial and temporal trend of these rapid events were carried out through remote sensing techniques,

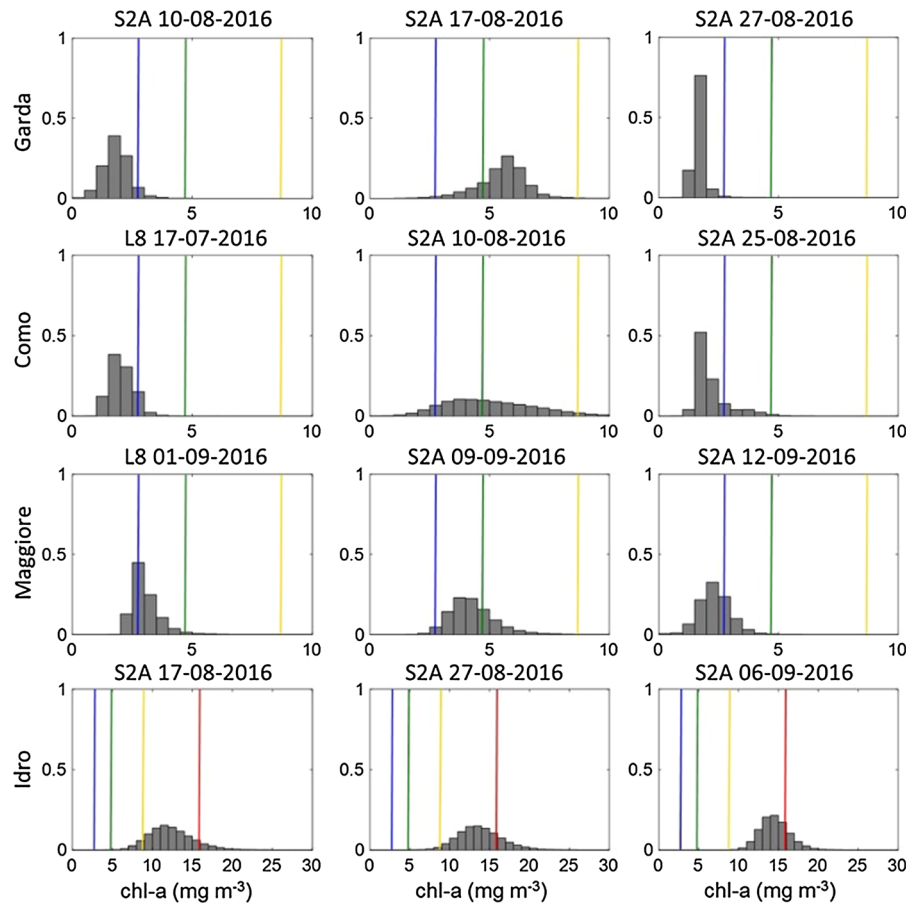


Fig. 8 Chl-a concentration (mg m^{-3}) distribution for each image: chl-a concentration on x-axis and fraction of total pixel on y-axis. Vertical lines indicate WFD boundaries for water

quality status classification based on chl-a concentration, respectively, blue: high/good status, green: good/moderate, yellow: moderate/poor, red: poor/bad (Wolfram et al., 2009)

while traditional limnological measurements alone would not have been enough to completely describe and, in some cases, even to identify these blooms. For example, in Lake Garda, the bloom was directly observed during radiometric field campaigns on 17th August 2016, but chl-a concentration measured in the same day in the Northern part of the lake would not allow to identify the bloom in the Southern area (Fig. 7). Considering chl-a concentration in all the stations used by different agencies for lake monitoring, values extracted by images in the lake showed that even if measures had been collected on these days in all stations, only some of them would have identified the huge variation in chl-a concentration, and none of them would have shown the highest concentration (Table 2). In the South-Western basin, the estimated chl-a concentration reached 8.91 mg m^{-3} , with a coefficient of variation of 0.87.

In Lake Como, the bloom was detected by in situ measurements on August 2016 11th and 17th in a Northern and in a Southern part, respectively: remote sensing products (even if no images were available between August 10th and 25th), differently from punctual measurements, allowed describing the spatial distribution of the bloom, not uniformly distributed in the lake.

In order to complete the analysis, and to understand the whole environmental conditions in which these rapid events can occur, meteorological data (from ARPA Lombardia monitoring stations next to the lakes), potentially affecting algal growth, were considered over the events time window.

As reported by Callieri et al. (2014), blooms of *D. lemmermannii* can be occasionally recorded even in deep subalpine lakes, as supported by nutrient pulses deriving from the mineralization of organic matter

deposited along the lakeshore and released through runoff during rainfall events. Nutrients arriving from the lake catchment area can stimulate phytoplankton growth, especially in oligo-mesotrophic lakes (Morabito et al., 2012), and, combined with a seasonal increase in water temperature, it would facilitate *D. lemmermannii* proliferation (Ollrik et al., 2012; Salmaso et al., 2015). Calm conditions are known to be advantageous for buoyant cyanobacteria which move toward the euphotic zone in response to reduced turbulent mixing (Jöhnk et al., 2008; Zilius et al., 2014).

In Fig. 9, precipitation, air temperature and wind velocity are shown in the period analysed for this case study. Data show that, for deep lakes, short-term weather conditions are not directly correlated with blooms phenomena but in the days of surface blooms, wind was relatively calm.

Discussion

Microscope analysis for species identification density and biomass measurement remains the standard method for estimating the species-specific biomass of phytoplankton in natural samples (Hillebrand et al., 1999; Harrison et al., 2015). However, it requires a high level of expertise, it is time consuming and the accurate estimates of diversity patterns depend critically on the taxonomic skills of the operator and on the counting effort (Olli et al., 2015). Moreover, phytoplankton bloom may contain both cyanobacteria and other algae as well as the presence of competitors, predators and parasites as factors influencing the population dynamics (Likens, 2010), thus making very challenging the species identification in particular from satellite-based technologies. Nevertheless, satellite data remain a valid tool to identify the extension and intensity of the bloom, helping environmental authorities and managers to set up the most suited control strategies. In fact, as showed from our results, satellite data allow a synoptic view of the area of interest with high frequency of observation. In particular, in the case of algal bloom events, traditional limnologic in situ measurements, which are carried out more rarely, could miss rapid events lasting only for a few days, while satellite optical data could catch bloom phenomena, except in case of clouds. On the other hand, the influence of cloud cover when using

Table 2 Chl-a concentration extracted from Lake Garda images and coefficient of variation (cv) over monitoring stations

Date	10/08/2016	17/08/2016	27/08/2016	cv
14	1.21	4.06	1.75	0.65
371	<i>2.01</i>	<u>5.42</u>	1.59	0.70
369	1.51	<u>5.04</u>	1.82	0.70
Padenghe	1.89	<u>6.42</u>	1.75	0.79
Salò	2.38	<u>5.76</u>	1.63	0.68

Cells are filled with colours based on WFD classes: bold if high, italic if good, bold italic if moderate, underline if poor

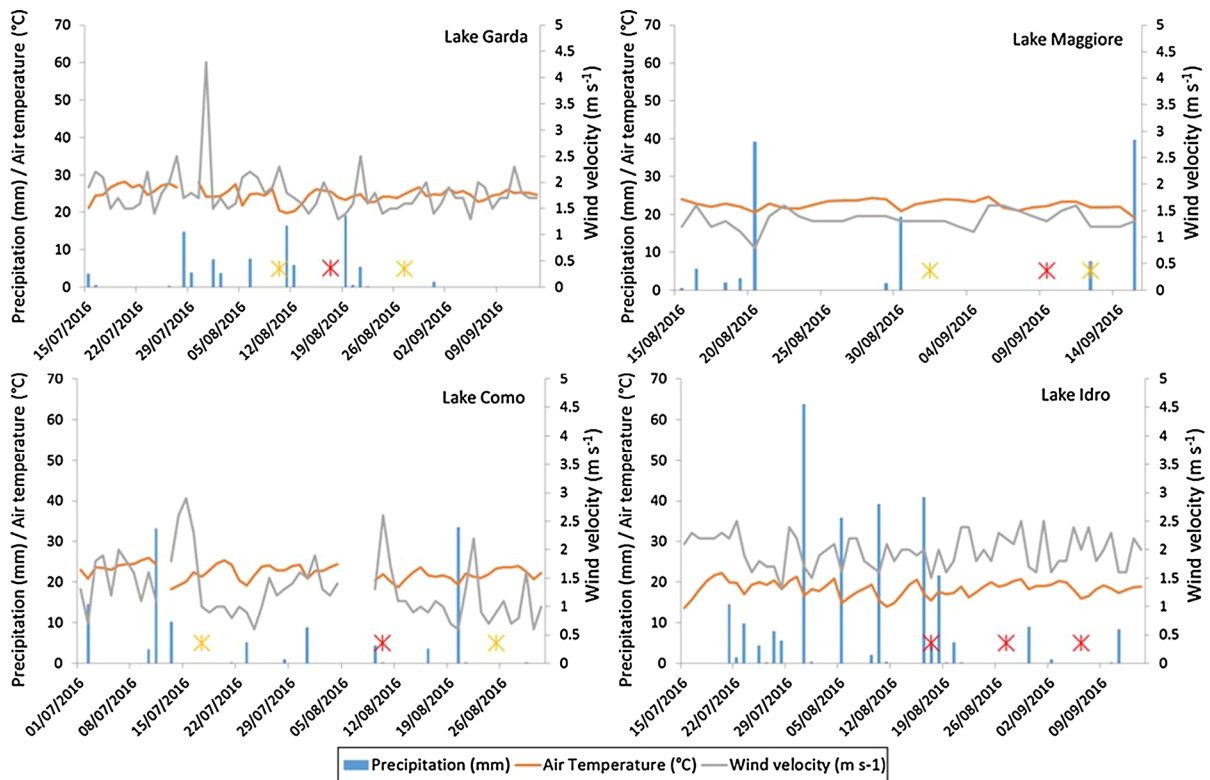


Fig. 9 Precipitation, air temperature and wind speed from some ARPA meteorological stations next to the lakes. Stars indicate images dates (red stars indicate the date of the image in which each bloom was detected)

satellite data could be significantly reduced by combining imagery acquired from several missions, such as L8 and S2A as shown in this study. In 2016, over the southern regions of Lake Garda, a total of 112 acquisitions were available by combining S2A and L8. Out of them, about half (55%, i.e. 62 images) were cloud free. By focusing to a shorter period (April 1st to September 30th), the percentage of cloud-free images increased up to 66% (39 images). For Lake Maggiore, cloud-free images were about 51% for the entire year and 54% for the same late spring/summer period. For both Garda and Maggiore, the longest consecutive period with images affected by cloud cover was November 2016, with 28 days without cloud-free images. Finally, the average lag between two successive cloud-free images was of 7 days.

The probability to find more cloud-free images is now enhanced with the availability of data collected by S2B, which was launched by ESA in March 2017. Revisiting time of combined medium resolution optical sensors L8, S2A and S2B in the study area is

of 2–3 days. Moreover, for water quality monitoring, the operational availability of data acquired by Sentinel-3 (S3) OLCI (Ocean and Land Colour Instrument) is of key importance. This mission provides high frequency of observation (i.e. revisiting time over our study area of about 1–2 days) with a spatial resolution of about 300 m. Although spatial resolution is reduced compared to S2 and L8 optical sensors, these data represent a principal source for water quality studies and monitoring. In particular, S3-OLCI offers 21 spectral bands in the visible–near-infrared regions of the electromagnetic spectrum to distinguish different types of phytoplankton, characterized by different pigments. L8 and S2 spectral bands alone do not allow the discrimination between cyanobacteria bloom and other algae bloom in terms of secondary pigments. In fact, these multispectral sensors do not carry a spectral band sensitive to, for example, the presence of phycocyanin and phycoerythrin, like S3-OLCI bands are. Again, a multi-source

approach is the most suitable for operational and full monitoring of lake water quality.

The satellite-based approach has then to be always integrated to in situ observations. For instance, an obvious limiting factor in the use of satellite data in the study area is the impossibility of detecting and quantifying deep phytoplankton blooms, such as those due to *Planktothrix* that in deep subalpine lakes usually occur at water depths around 20 m (Salmaso, 2005). In this case, the absorption and backscattering properties of phytoplankton cannot be detected due to lower penetration of light through the water column, since light is completely absorbed by the water column and the electromagnetic signal returning back to the satellite sensor is too weak to detect the presence of an algal bloom.

Finally, to use satellite data to obtain robust and reliable quantitative products of water quality parameters over lakes district (e.g. subalpine lakes), a strict requirement is to have a robust atmospheric correction method, validated over lakes with different trophic status. To this aim, we relied on in situ measurements of AOT from photo-radiometers belonging to the CIMEL network, which were exploited for correcting the imagery from the atmospheric effects. Nevertheless, further investigations are necessary to both increase the number of match-ups and to evaluate uncertainties due to radiometric measurements gathered in the field.

Conclusions

Since the latest satellite sensors used (L8-OLI and S2A-MSI) have not primarily designed for observing the optical properties of water, this study shows that they have suitable characteristics to support environmental monitoring in subalpine lakes. Moreover, the obtained results confirm the robustness of physic-based approach based on atmospheric correction and bio-optical modelling inversion, to retrieve chl-a concentration.

The results show that combining remote sensing with in situ measurements can help to monitor phytoplankton blooms for the WFD in subalpine lakes. In this study, high spatial resolution products were useful to recognize spatial pattern of cyanobacterial blooms, while combining products from the two different satellites (and in the future integrating their

time series with S2B and more frequent S3 products) allowed assessing on time the occurrence of these events. If these medium spatial resolution multispectral sensors bring a great advantage for the analysis of trend at spatial scale, their integration with data acquired by sensors with daily overpass and improved spectral resolution, such as OLCI on S3, could allow a continuous monitoring of water quality parameters and phytoplankton blooms for the entire subalpine lake district.

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